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Ungulate Habitat Restoration In Elk Island National Park

by

Colin John Van Bostelen

A thesis submitted to the Faculty of Graduate Studies and Research in partial fulfillment of the requirements for the degree of Master of Science

in

Land Reclamation and Remediation

Department of Renewable Resources

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Dr. David S. Chanasyk

Date: May 30, 2003

ABSTRACT

Six native grasses were seeded to disturbed soils at Elk Island National Park to restore wildlife habitat. Two seeding treatments and two fencing treatments were evaluated for three growing seasons. Ground cover, density, biovolume, biomass and species richness were assessed. Overall fencing had a significant impact on the vegetation parameters assessed but broadcast versus drill seeding did not. There was no intervention between the fencing and seeding treatments. Wildlife heavily utilized the research area and fencing prevented the impact of that utilization. The seeded wheatgrass species and fringed brome showed potential for revegetation; green needle grass and nodding brome either did not survive or did so in very small numbers and did not increase with time. The seed mix did not prohibit invasion by non-seeded species. Many non-seeded weed and desired species grew in the sites.

"I became insane, with long intervals of horrible sanity"

Edgar Allan Poe

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Well I can't believe I'm writing this. It must mean that I've finally come to the end of a long and arduous journey. I can honestly say that this has been one of the toughest things I have ever tried to do in my life. For me school and working for a living have always been at opposite ends of the spectrum. When I was in school all I could ever think about was getting out of school and working and when I was working at various menial jobs all I could think about was going back to school. I'm sure after a few months have passed I'm going to find myself wanting to learn something new again. I can remember when I was in grade 9 (1979) and I was called into the guidance counsellor's office at school. The counsellor could not understand why I was unsure of what I wanted to do for a living. Well here it is twenty some years later and I feel like I'm still sitting in that chair wondering what to do with my life. I hope that I will never give up the hunger for learning new things, whether it is in school or out in the real world.

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CHAPTER 1. INTRODUCTION

1.1 ECOLOGICAL RESTORATION

Since its inception, ecological restoration has been defined by various authors and societies. The Society for Ecological Restoration (SER) has defined ecological restoration as the process of assisting the recovery of an ecosystem that has been degraded, damaged or destroyed (SER 2002). This is rather simplistic compared to their original definition adopted at the society's commencement in 1990. Ecological restoration was then defined as the process of intentionally altering a site to establish a defined, indigenous, historic ecosystem. The goal of this process is to emulate the structure, function, diversity and dynamics of the specified ecosystem (SER 1990). A simplistic definition that works best for this project comes from Cairns (1993) and it states that ecological restoration is the reestablishment of the structure and function of ecosystems to predisturbance conditions.

As time has progressed, defining ecological restoration has encompassed a broader scope of variables. A modern approach to classifying the fundamentals of ecological restoration does not only require technical analyses but must take into account ethical subject matter as well (Higgs 1997). Ethical subject matter comprises historical, social, cultural, political, aesthetic and moral contexts (Higgs 1994). Higgs (1997) suggested reconsidering the term restoration which connotes a fixed, historic and static notion of ecosystems and has proposed replacing it with regeneration, which gives open recognition to the tentative, developmental quality of sensitive human engagements with ecosystems.

Ecological restoration is a relatively new academic discipline that has evolved due to increased anthropogenic disturbances. As the world population multiplies there is an increasing demand placed on natural resources which eventually leads to habitat degradation. Young (2000) believes that 50 years from now the majority of the world's habitats and species will either be destroyed or on their way to recovery from a degraded state due to the influx of a larger population base. As we approach this impending crisis our attentions have to focus on the conservation and restoration sciences.

Young (2000) states that restoration ecology is a subset of conservation biology but the long term future of conservation biology is restoration ecology. He further states that conservation biology is the science which attempts to reduce habitat and biodiversity loss whereas restoration ecology is the science of habitat and biodiversity recovery. There are inherent problems which can come about if habitat and biodiversity recovery plans are not implemented after a disturbance. A paramount concern of leaving restoration to natural processes is the time that it takes to return to these states. Estimates of time scales necessary to complete restoration are often too optimistic due to process rates of both abiotic transformations and species turnover (van Diggelen et al. 2001). Dobson et al. (1997) proposed that the time required for restoration can span from decades to centuries and redevelopment of advanced communities may take a millennium or more. Restoration ecology provides a powerful suite of tools for speeding recovery of degraded lands, however, this process is not possible unless specific problems are identified and overcome by artificial intervention that mimics natural processes (Dobson et al. 1997).

The current challenge of ecological restoration is to ensure that structure and function recover, which will allow a full range of species to find their niches (Jordan et al. 1987). To achieve restoration success, ecologists should glean knowledge from past accomplishments and utilize information from them as a template for future projects instead of relying on investigation of underlying principles and mechanisms (van Diggelen et al. 2001). However, ecological restoration cannot only focus on past experiences but should also comprehend how restoration will affect the relationships between people and ecosystems in the future.

Size of disturbance is another critical parameter requiring consideration prior to restoration. Hierarchy theory predicts that habitat use is influenced by the interaction of factors at multiple scales from the microsite to the landscape and that higher order factors impose constraints at lower levels (George and Zack 2001). Consideration of only the landscape context might make a site unsuitable for a species even if vegetation structure and composition are restored (Morrison 2001). Van Diggelen et al. (2001) suggest restoration (return to predisturbance conditions) at the landscape scale becomes too costly and reclamation (returning disturbed land to former or other useful purpose) may become

the realistic option. Kentula (1997) states rehabilitation (return land to a useful purpose) may be another option if only a portion of the landscape requires restoration, for example, riparian habitat.

The establishment of clear and well defined goals is needed to facilitate any restoration project. Ehrenfeld (2000) suggests goals must be developed exclusively for each project and must realistically recognize that functionally created ecosystems are not exact replacements of the originally disturbed natural ecosystem. Restoration projects should not begin until disturbed sites have been thoroughly investigated. An understanding of soils, flora, fauna and the end land use goal for the area to be restored is essential prior to undertaking restoration. This includes reestablishment of former functions, characteristic species, communities and structure (Grootjans et al. 2001, Pfadenhauer 2001). Success can be measured by clearly defining target communities and/or target species in the restoration project (Hobbs and Norton 1996, Pfadenhauer and Grootjans 1999, Bakker et al. 2000). Targets serve as end land use objectives and proximity to that target is used to verify success. The amount of time necessary to deem a project successfully restored may take decades or longer. It is imperative that monitoring accompany every restoration project to determine if applied techniques are working or if further intervention is required.

1.1.1 Habitat Restoration

The use of native plants in revegetation coupled with influences from wildlife present restoration ecologists with several challenges. Due to the complex dynamics of ecosystem processes in disturbed areas, habitat restoration projects become multifaceted endeavors necessitating comprehensive recovery plans. Revegetating disturbed areas in National Parks with native plants and the effects surrounding wildlife populations can have on these projects are present day concerns, which initiated this research.

Using native plants for revegetation will decrease the introduction of non-native plants and assist in maintaining the integrity of existing native plant communities. Native plants also provide habitat for wildlife which is the end land use goal for this project. These sites will serve as a reference point for future revegetation projects located in the

park. If the native grass mix flourishes, it will be utilized on other disturbed areas that may arise from park activities.

Prior to revegetation, soil in the disturbed site must be assessed. Determining soil type, potential for erosion and organic matter content are a few of the relevant issues that need to be addressed before plant species selection can commence.

What to plant, when to plant and how to plant are three pertinent questions that must be answered after soil assessments have been completed. Native plants existing in ecoregion(s) surrounding the park address the first question of what to plant. Park policy prevents the introduction of non-native plant species into the park's plant communities, therefore only native plants can be used. This mandate prevents utilizing any agronomic species in revegetation seed mixes. For example, even when the potential for erosion is high in a disturbed area, the use of an annual agronomic species as a cover crop is strictly prohibited. Parks Canada's mandate to use only native plant species for revegtation ensures ecological integrity of native species will be met.

The selection of individual native species and consideration of their growth characteristics and adaptations is critical for revegetation success (Ducks Unlimited 1995, Morgan and Collicutt 1995). Availability can sometimes become a limiting factor when deciding species for the seed mix. A greater number of species in the seed mix ensures biodiversity goals set out in the park's management plan are realized. Increasing the variety of seeds in the mix hypothetically increases habitat diversity which in turn increases selection for animal diversity. Species mixes also provide several advantages over monocultures because each species in the mix is adapted to one or more of the different environmental conditions and stresses that may occur in the disturbed site (Vogel 1987). In addition, they should be suited to diverse ecological, physiological and reproductive life cycles. For example, a mix of deep rooted and shallow rooted grasses may be more efficient at accessing moisture and nutrients in the soil than a monoculture with uniform rooting depth (Vogel 1987). Tolerance to drought, pH and salinity are other examples of plant suitability criteria requiring consideration before seeding. In this project, selection of plants is also dependent on wildlife. Plant productivity, palatability, nutrition and response to grazing are additional factors that require careful planning when choosing a seed mix. A seed mix consisting of species that can acclimatize to varying

scopes of environmental conditions and impacts from wildlife will benefit habitat restoration projects in disturbed areas and provide them with a greater potential for success.

There are several advantages and disadvantages to using native species for reclamation on disturbed sites. Native species have evolved over many years and therefore are adapted to local conditions. When native plants are seeded in disturbed sites, they often require less labor and expense to maintain once established. Native plant species assist in preventing weeds from dominating revegetated sites and do not require long-term management, such as fertilization (Pelech 1997). They are well adapted to harsher environmental conditions and are long-lived species that have genetic diversity, which ensures biodiversity (Naeth 1999). Biodiversity assists native plant communities by making them potentially more resistant to disease and stress (Pelech 1997). Native plant species have diverse dates of maturity, an ability to adapt to grazing and are capable of remaining palatable and nutritious late into the growing season which makes them an excellent food source for ungulates (Trottier 1992).

There are some disadvantages associated with selecting native plant species for revegetation. They are often slower to germinate and establish than non-native species (Pelech 1997). Reduced emergence and establishment rates are precursors for other setbacks which can impede revegetation time frames. Takyi and Islam (1984) concluded that cultivated grasses developed plant cover for erosion control faster than native grasses in the eastern slopes. A final consideration requiring attention is the seed source. In some instances disturbed areas cannot be seeded with a sufficient variety of native species indigenous to the region. The lack of a regular, commercially available volume of species at an affordable price precludes native species from being selected for revegetation (Sutton 1975, Richards et al. 1998).

After suitable native plant species have been chosen for the project, time of seeding needs to be addressed. It has been a generally accepted practice to seed prior to the season that receives the most precipitation (Vallentine 1989). Seeding can be completed in spring, early summer or fall. The optimum season of seeding depends upon the time the disturbance occurred. Leaving exposed soil for an extended period of time after the disturbance renders it susceptible to erosion and invasion from non-native

species. For example, if the area became disturbed in the winter months, seeding should occur in the spring. Conversely, if a disturbance occurred in August, fall seeding would suffice. The time of seeding is also dependent on species. Cool season species should be seeded in early spring (April to mid-May) or in the late fall (after mid-October) whereas warm season species should be seeded mid-May to mid-June (Vogel 1987, Hardy BBT Ltd. 1989, Gerling et al. 1996).

Planting method is dependent upon seeding techniques available to restoration ecologists. These techniques incorporate seeding methods, rates and depths. Each one of these factors is dependent on the other. For example, if an area needs to be drill seeded, that seeding method will dictate the seeding rate and seeding depth for species comprising the mix. Perhaps the greatest limiting factors for seeding are accessibility and slope.

The two most widespread seeding methods used are drill and broadcast (Vogel 1987, Gerling et al. 1996). Drill seeding controls the rate of seeding and distributes the seeds uniformly, deposits varying seed sizes to varying seed depths, makes good soil/seed contact which assists the seed in germination and establishment and protects the seed from erosion and predators. Finally, it is cost effective because it uses less seeds ha^{-1}. However, when disturbed sites cannot be accessed by a drill, an alternative seeding method is required, such as broadcast seeding. Accessibility can be hindered by wetness, steep slopes, rough terrain and soils that have a high stone content. Broadcast seeders usually have bigger openings to disperse seed negating the use of a carrier.

Some disadvantages associated with drill seeding are that hairy seeds or those with awns can plug the seeder cones requiring a carrier to facilitate movement of seed through the drills. Drill seeding creates rows which contradict the idea of a natural environment. The increased amount of seed required for broadcast seeding coupled with lower establishment and increased costs are drawbacks. Broadcast seeds are also highly susceptible to predation from wildlife and erosion from wind and water.

Habitat restored with adequate native vegetation can still fail to support species due to insufficient space. Taylor and Taylor (1979) stated that several animal species exhibit a dynamic spatial distribution of populations, which suggests they may require more space than they occupy at any given time. The effectiveness of restored habitat is

also dependent upon its capacity to sustain a functioning population and therefore; should provide ample space for populations to periodically relocate as resources are depleted (Smallwood 2001). Wildlife habitat is not static, but continually changes over time with natural and anthropogenic disturbances, and therefore, ecological processes that contribute to those dynamics must be maintained or substituted in the recovery plan (Engstrom et al. 1999, George and Zack 2001). To successfully restore habitat for wildlife, understanding the effect of wildlife populations across spatial and temporal scales is crucial.

Wildlife populations can dictate the success of a restoration project. If wildlife numbers exceed the ecological space available to support functionally significant demographic units ecological restoration is difficult to achieve (Smallwood 2001). Animal species must have access to additional habitat if degraded areas are to be successfully restored. If wildlife populations are excessive and similar habitat to the previously disturbed area is not accessible, restoration becomes hindered and may not take place at all. There is nothing restoration scientists can do when the carrying capacity of wildlife exceeds the habitat requiring restoration (Smallwood 2001). If population numbers surpass carrying capacity, wildlife should be excluded until the project has been completed. To ensure habitat restoration success the density of wildlife inhabiting the landscape which incorporates the disturbed area must be known.

Restoration projects cannot be effective without considering the area necessary to support spatially constrained wildlife species. Without consideration of spatial capacities of representative wildlife residing in and adjacent to disturbed areas, restoration efforts may fail. Restoration projects are jeopardized when space becomes the limiting factor for wildlife. Disturbed areas limiting access to useable habitat can force wildlife to utilize disturbed areas even if restoration has not been completed. Herbivory in the project site prior to the establishment of vegetational species can inhibit or prevent revegetation success.

The time required to restore a disturbed habitat varies with site. Time is a factor that has two functions relevant to restoration. Restoration is dependent upon temporal ecological processes that naturally occur in the disturbed area. For example, if revegetated areas receive little or no precipitation following seeding, restoration is

unlikely. Time must also be viewed in light of wildlife visiting the disturbed site. Granting access to wildlife during revegetation can delay the time necessary for restoration. Wildlife will inhibit plant emergence and establishment, lengthening the time to complete the restoration process.

A multitude of factors affect successful habitat restoration. The selection of suitable native plant species and the associated tasks of seeding them intertwined with wildlife populations and the role they play spatially and temporally in disturbed areas are important issues that must be addressed to achieve restoration success.

1.2 HABITAT RESTORATION CONSIDERATIONS IN ELK ISLAND NATIONAL PARK

This research project required the amalgamation of four key components: National Parks, revegetation, ungulates and fencing to successfully restore habitat for wildlife. There are times when anthropogenic activities create disturbed areas in National Parks and in the case of Elk Island National Park (EINP) two such sites existed. Park staff decided creating habitat for ungulate species residing in the park should be the end land-use goal for the project. Overabundant ungulate populations in EINP and the relatively small landscape they occupy dictated the need for habitat restoration. An examination of the results collected from this project will aid in formulating revegetation policy and may be integrated into future EINP management plans.

The first component in this research is National Park guidelines in relation to revegetation projects. As stated previously, park guidelines must be strictly followed when revegetation projects are conducted in National Park settings. Since EINP is in a transition zone between the Lower Boreal Mixedwood Ecoregion and the Aspen Parkland Ecoregion, plants indigenous to these regions are required for revegetation.

Revegetation is a restoration component which can prevent further damage to areas denuded of vegetation. Soil erosion and invasion from non-native species can severely alter the biodiversity of an ecosystem. Revegetating disturbed habitat will return naturally occurring ecosystem processes to disturbed areas. Once vegetation becomes established, revegetated habitat will provide food, cover and home range for wildlife.

Restoration of quality habitat is obtained by planning and providing for the needs of desired wildlife species (Vogel 1987). Ungulate populations affecting this research project were elk (*Cervus elaphus*), moose (*Alces alces*), plains bison (*Bos bison bison*), mule deer (*Odocoileus hemionus*) and white-tailed deer (*Odocoileus virginianus*). Current population levels for these species are above the recommended carrying capacity (Cool 2003). Wildlife activities from each of these species can impact revegetation projects and these impacts can hinder or even prevent newly seeded vegetation from establishing. Utilization of habitat by ungulates can arise from bedding, grazing, trampling, wallowing and excretion. Each of these ungulate interactions has the potential to hamper the revegetative process. Bedding on sites will crush emerging plants and delay establishment. Grazing of established vegetation will force the plant to use carbohydrate reserves because photosynthesis will not produce enough energy for the plant and this may eventually kill the plant. Trampling of newly seeded disturbed areas can lead to a number of deleterious impacts. Hoof action may force seeds to a depth that will prevent germination and establishment. Compaction of the soil can exacerbate the situation further. Wallows remove all of the existing vegetation and displace drill and broadcast seeded seeds. Dung from ungulates may contain seeds of non-native plants thus creating competition between seeded and non-seeded species.

Fencing is a key management option that must be considered in habitat restoration when wildlife populations limit revegetative success. Since wildlife interactions can be detrimental in a revegetation project, fencing newly seeded sites must be considered as a compulsory management tool. Fencing will give seedlings an opportunity to emerge and establish thus impeding invasive species from dominating disturbed sites. In some instances, fencing will not be possible due to accessibility and cost, and alternative options will have to be pursued.

To ensure the success of a revegetation project there are a plethora of factors that must be contemplated before it can even start. Soil assessments must be the first priority when disturbed areas require revegetation. After the soil is analyzed suitable plant species that match the soil conditions and are also palatable or unpalatable to wildlife can be selected. Wildlife in the vicinity of the project should also be assessed to see if there

activities have the potential to negatively impact revegetation. If this is the case, fencing should be used to prevent access to the site in the early stages of the restoration process.

1.2.1 National Parks Habitat Restoration Guiding Procedures

At present there is limited published literature on revegetating degraded habitat in national parks. National parks are considered representative areas of Canada that are untouched for the benefit of present and future generations. However, due to intensive activities inside national parks, disturbed areas have emerged. Reclamation projects are now being undertaken in the national park setting to restore areas which were subjected to human intervention. Research in protected areas, especially national parks, can serve as a benchmark to measure direct effects of human impacts on biodiversity loss and valuable data collected will assist restoration ecologists on how to proceed with restoration efforts in the future (Sinclair 1998, Sinclair et al. 2002). Utilizing native vegetation to restore these areas will provide additional information for future revegetation plans, as well as adhering to policy.

Native plant policy dictates the use of native plant species for revegetating national parks. In the United States, restoration ecologists are using native seed for revegetating rangeland to maintain or restore the genetic and ecological integrity of native ecosystems in national parks (Richards et al. 1998). Prior to any restoration efforts in a national park certain rules and regulations must be followed.

Policy and guidelines on plant species suitable for revegetation dictate habitat restoration procedures. Restoration projects conducted in national parks are subjected to a hierarchical chain of legislation, policies, guidelines and planning. The flow of regulating authority commences with the National Parks Act and is followed by Parks Canada Guiding Principles and Policies, individual Park Management Plans, Ecosystem Management Plans, Implementation Plans, Operational Plans and Monitoring (Parks Canada Agency 2000). These various steps are the basis on which every restoration project occurring in a National Park setting must adhere to.

1.2.1.1 The National Parks Act

The National Parks Act and its associated regulations are the foundation on which all subsequent levels of policy and planning are based on. The Act was first ratified in 1930 and has been amended several times since its enactment (Blyth and Hudson 1987). The underlying premise of the National Parks Act is to ensure that the people of Canada will have access to natural areas and that they will remain unimpaired for future generations. Policy direction for the implementation of the National Parks Act is set out in Parks Canada Guiding Principles and Operational Policies.

1.2.1.2 Parks Canada Guiding Principles and Operational Policies

Parks Canada Guiding Principles and Operational Policies is the next level governing activities in national parks. In 1988 an amendment to the National Parks Act stated that the maintenance of ecological integrity must be the first consideration in management planning. The central theme of this guiding principle is to ensure each national park uses a strategic framework to sustain integrity of individual ecosystems.

Park policy stipulates that natural processes will be managed with minimal interference; however, active management is permitted when the structure or function of an ecosystem has been seriously altered and manipulation is the only alternative available to restore ecological integrity (Parks Canada 1994). Policy also recognizes that partnerships with universities are important and how much these institutions can contribute towards research and monitoring restoration projects within national parks (Parks Canada 1994). To meet the goals set out by Parks Canada principles and policies, every National Park must prepare a five year management plan.

1.2.1.3 National Park Management Plans

The term Park Management Plan has been retained by the Parks Act, but it has been changed to a new format titled Ecosystem Conservation Plans and incorporates an Ecological Integrity Statement (Parks Canada Agency 2000). Park Management Plans must contain management objectives in adequate detail to indicate how a park will protect the natural aspects of its region and in keeping with these objectives, plans will specify the type and degree of resource protection and management needed to assure the

ecological integrity of the park. Management plans are reviewed every five years and can be retabled with any amendments pertinent to maintaining the ecological integrity of the Park. Park management plans provide the framework for further detailed sub-plans concerning ecosystem management and are individually suited to distinct regions in which each park is located.

When disturbances arise in a National Park and research confirms that intervention is necessary to restore the damaged ecosystem, management must be guided by the establishment of clear, practical and measurable objectives consistent with the park management plan (Parks Canada 1994). Restoration efforts have to be based on scientific research, use techniques that duplicate natural processes as closely as possible and be carefully monitored. All practical efforts must be made to prevent the introduction of non-native plants into disturbed areas (Parks Canada 1994).

1.2.1.4 Elk Island's Ecosystem Management Plan

Elk Island's ecosystem management plan receives its direction from the park's Implementation Plans and is carried out in their yearly Operational Plans. Monitoring of yearly operations conducted inside the park and reported every two years in the State of the Park Report. Additional input dealing with the complex interactions and dynamic processes of the park's ecosystems is provided by the Science Advisory Committee (SAC). SAC provides advice on science based research, monitoring and management initiatives to ensure management decisions are based on sound ecosystem based management information (Cool 1999).

As of 1999, management plan directives for EINP were to restore landfill sites with a native seed mix (Cool 1999). The following year, objectives of the Canadian Environmental Assessment Act plan were to minimize creating disturbed areas in the park. Where disturbed areas were created, seeding native plant species was used to minimize noxious weed invasions (Chapman 2000). It is Parks Canada policy to use only appropriate native plant species in all of Elk Island's reseeding efforts (Chapman 2000).

In 2001 SAC outlined details for an Ecological Restoration Plan (ERP). This plan dealt with working in disturbed areas and included clear objectives, potential maintenance activities, a detailed monitoring program, performance measures to

determine success and a budget (Parks Canada Agency 2001). Essentially this plan entails restoring anthropogenic disturbances with native plant species when a potential for invasion of non-native plants exists and reestablishment of native species is unlikely due to loss of seed sources. Maintenance is also incorporated into the plan and dictates that undesirable plant species should be removed, herbivore use should be reduced or prevented and soil erosion should be reduced or prevented until restoration objectives have been reached (Parks Canada Agency 2001). Monitoring was also incorporated into this program to ensure stated goals would be achieved.

At the present time the ERP operational plan is quite similar to the goals set out by this research project. This project was seeded with native seeds due to a disturbance created from anthropogenic activities. Two seeding treatments, as well as two fencing treatments were monitored for three years to assess restoration efforts.

1.2.2 Revegetation in National Parks

The majority of revegetation history in the literature comes from the United States. Revegetation efforts date back to the 1890s when rangeland was reseeded to prevent land degradation resulting from past stocking practices with domestic animals and the abandonment of cultivated former rangeland (Call and Roundy 1991). Early revegetation efforts relied heavily on agricultural practices and were largely unsuccessful because of seed availability and poor site preparation (Stoddart et al. 1975). Projects dating back to the 1950s, 1960s and 1970s focused on stabilizing rangelands and watersheds with non-native species because they were easily produced and established, were grazing tolerant and state and federal legislation mandated it. (Meeuwig 1960, Keck 1972, De Puit 1986). In the 1980s, disturbed areas were starting to be revegetated with native species, but there was difficulty in producing, harvesting and sowing them (Booth 1987). Some revegetation projects were conducted using non-native native plant crosses and they played a crucial role in stabilizing rangelands (Carlson and McArthur 1985, McArthur 1988). The problem with seeding non-native species is that as they become established, native forage species used by ungulates may receive such a small share of available resources that they are able to persist only at very low population levels (Cody 1986). The 1990s have seen National Parks mandate the use of native species for all

revegetation projects on disturbed areas within park boundaries. The future will see increased use of native species for improving wildlife habitat because wildlife biologists presently consider habitat degradation and loss to be the most important issues threatening wildlife (Flather and Hoekstra 1989).

Revegetation projects conducted in the past have either been drill seeded or broadcast seeded. Ocumpaugh et al. (1996) studied switchgrass recruitment from broadcast seeding versus seed fed to cattle. Restoration of disturbed areas in Yellowstone and Glacier National Parks have been broadcast seeded, followed by hand-raking or dragging the surface to bury seed and improve seed soil contact (Majerus 1997). Most of the drill seeding literature focuses on agronomic species such as wheat and rye grass. Presently there is very little literature comparing the growth of drilled and broadcast seeded native species in revegetation projects. Literature involving these two seeding techniques in revegetated sites set in a national park is almost non-existent. Pitchford (1999) determined drill seeding to be more effective than broadcast seeding at Elk Island National Park. She found that drill seeding is more advantageous than broadcast seeding because it permits better seed soil contact and prevents wind erosion of seeds.

Research by Lamb (1998) assessed revegetation success of disturbances in Glacier National Park, British Columbia. Although her research dealt with revegetation in a National Park, it focused on agronomic species, not native species, and did not assess fencing or ungulate interactions. Roundy et al. (1997) recommended that revegetating with native species follows the increasing emphasis on conservation goals for rangelands and protected areas. Kay (1997) has researched ungulate herbivory on willows (*Salix* sp.) in Yellowstone National Park and the fencing of some sites; however, the use of native grass species was not addressed.

1.2.3 Potential Ungulate Effects on Revegetation in National Parks

The current literature has no data on effects of ungulates on revegetated sites; however, there is an abundance of literature on their effects on established rangeland (Hudson and Nietfeld 1985, Frank and McNaughton 1993, Singer 1995, Hobbs 1996, Bork et al. 1997, Detling 1998). Stress on revegetated sites can arise when ungulates trample newly seeded sites and emerging plants, graze emerging plants and bed in newly

seeded areas. Each of these activities has the potential to hinder or prevent successful revegetation. When ungulates place stress on emerging plants, future revegetation projects may need to consider fencing of the entire site prior to seeding.

Ungulates mainly affect habitat by grazing, wallowing, trampling, bedding, defecating and urinating. Ungulates also select different vegetation types, plant species and plant parts while grazing and browsing. In the management of revegetated sites animal use may be the single most important variable requiring attention. Regeneration of preferred species can be seriously hampered or even prevented by ungulate activities in newly revegetated areas (Van Wieren 1995). Ungulate dung may also contain undesirable non-native seeds and subsequent competition from these species may hinder the establishment of native species (Janzen 1984).

Vinton and Hartnett (1992) found that tillers repeatedly grazed in unfenced prairie had lower relative growth rates and accumulated significantly less biomass than ungrazed tillers. Although grazing the previous year resulted in more tillers initiated per m^2, these tillers suffered higher mortality later in the season than tillers that had not been grazed. Observations also showed that grass patches grazed by bison early in the season were regrazed later in the season rather than creating new grazed patches (Vinton 1990). Fenced areas that had the fencing removed in mid-summer to allow later grazing were completely avoided by bison. Thus, a tiller that is grazed early may or may not be grazed later in the season and tillers that are not grazed early are more likely to remain ungrazed throughout the season (Hartnett 1989).

Selective use of plants by herbivores as a result of palatability differences or frequency-dependent defoliation can influence the abundance of other plant species and result in changes in plant species diversity (Grime et al. 1987). In prairies, grazed patches are stable and used repeatedly from year to year, resulting in the development of a patchy grazing mosaic at the landscape level (Hobbs et al. 1991). Heterogeneity in bison grazing will likely affect ecosystem processes by altering patterns of productivity over the landscape (Vinton et al. 1993). For example, intensively grazed patches may result in waves of increased production around patches, via the competitive release of uneaten plants (Franco and Harper 1988).

Polley and Collins (1984) conducted research on wallows created by bison and observed that wallows contained considerably more clay than soils of the surrounding prairie. They determined topographic depressions in wallows coupled with the higher clay content of their soils increased moisture in wallows compared to adjacent prairie throughout the growing season. Their study established that vegetation found inside actively used wallows was consistently different from the vegetation surrounding the wallows. In each case, these vegetational differences corresponded to differences in soil texture, soil moisture and soil pH along the topographic gradient from wallow to prairie. Finally, they noted that the absence of bison disturbances allowed the establishment of adjacent prairie species in the wallows and as succession continued it eventually displaced species initially found in the wallows.

Trampling by large ungulates such as bison affects species composition. Van der Valk (1975) reported that microrelief created by large ungulate trampling altered Iowa fen vegetation and allowed certain weedy species to become established.

Ungulate interactions and the potential impacts they can incur in revegetation efforts have not received much attention. When these two parameters are coupled with a national park setting the research literature is non existent. Research in this area will provide national parks with invaluable data for future restoration projects.

1.2.4 Fencing Restoration Projects in National Parks

Restricting access through the use of exclosures is generally recommended to restrict access of grazing animals to areas where grazing is undesired (Osko et al. 1993). Osko et al. (1993) concluded that fencing revegetated rangeland in the Aspen Parkland of Western Canada was better than using chemical repellents to protect vegetation from grazing. Fencing revegetated sites from livestock is common but using fence to control ungulates is not. Protection from livestock grazing on revegetation projects can be easily accomplished using five strand barb wire fences; however, exclusion of big game is almost never achieved due to the high costs of fencing (Austin et al. 1994).

Studies in Britain have focused on using electric fences as protective measures to prevent non-native muntjac deer (*Muntiacus reevesi)* in Monks Wood National Nature Reserve from causing significant damage to coppice regrowth (Cooke and Farrell 2001,

Cooke and Lakhani 1996). The electrical fence used in their project only had one strand of charged wire and nominally prevented deer from browsing. Cooke and Lakhani (1996) realized that although electric fences provided a significant degree of protection for regrowth, they are rarely successful and can fail to give acceptable results in conservation efforts.

A study conducted by Kittredge and Kelty (1992) in New England utilized five strand electrical fence and five strand barb wire fence to prevent white-tailed deer (*Odocoileus virginianus*) from browsing Red Oak regeneration. Prior to establishing the deer enclosures, understory vegetation was prevented from growing above 100 cm in height by deer browsing.

1.3 RESEARCH GAPS ON HABITAT RESTORATION IN NATIONAL PARKS

At present, published research addresses two or three of the major components for restoring habitat, however, there has been minimal to nonexistent research conducted that covers all of them. Current literature does not address the concept of fencing seeded areas in national parks to prevent native ungulates from degrading revegetated sites. Given the opportunity to emerge and establish, native grasses in an enclosure may prevent habitat degradation and in turn promote a quicker restoration of the damaged habitat.

As stated previously, restoration ecologists should refer to past restoration projects and use them as a template for future restoration projects. In the case of Elk Island National Park, restoration of disturbed areas required park staff to only broadcast seed a native species mix. There are no guidelines other than this to restore degraded areas that may arise from anthropogenic activities occurring in the park. The goal of this research was to evaluate different seeding and fencing treatments in revegetated areas. Examination and interpretation of data collected will be used to construct a template that park staff can utilize for restoring future disturbed areas in EINP.

1.4 REFERENCES

Austin, D.D., P.J. Urness and S.L. Durham. 1994. Impacts of mule deer and horse grazing on transplanted shrubs for revegetation. J. Range Manage. 47(1):8-11.

Bakker, J.P., A.P. Grootjans, M. Hermy and P. Poschold. 2000. How to define targets for ecological restoration. Appl. Veg. Sci. 3:3-6.

Blyth, C.B. and R.J. Hudson. 1987. Elk Island National Park: a plan for the management of vegetation and ungulates. Environment Canada, Parks. Ottawa ON. 343 pp.

Booth, D.T. 1987. Diaspores of rangeland plants: ecology and management. In: Proc. Symposium on Seed and Ecology of Rangeland Plants. G.W. Frasier and R.A. Evans (eds.). U.S. Department of Agriculture, Agricultural Research Service. National Technical Information Service. Springfield VA. Pp. 202-211.

Bork, E.W., R.J. Hudson and A.W. Bailey. 1997. *Populus* forest characterization in Elk Island National Park relative to herbivory, prescribed fire and topography. Can. J. Bot. 75:1518-1526.

Cairns, J., Jr. 1993. Ecological restoration: replenishing our national and global ecological capital. In: Nature Conservation 3: Reconstruction of Fragmented Ecosystems. D.A. Saunders, R.J. Hobbs and P.R. Ehrlich (eds.). Surrey Beatty and Sons. Sydney AUS. Pp. 193-208.

Call, C.A. and B.A. Roundy. 1991. Perspectives and processes in revegetation of arid and semiarid rangelands. J. Range Manage. 44:5343-549.

Carlson, J.R. and E.D. McArthur. 1985. Symposium: range plant improvement. In: Proc. Selected Papers 38th Annual Meeting Society for Range Management. February 11-15, 1985. Salt Lake City UT. Society for Range Management. Denver CO. Pp. 107-220.

Chapman, R.J. 2000. Elk Island National Park three year CEAA plan 2000-2003 (including Code of Good Practices Environmental Assessment Guidelines). Unpubl. Rep. Fort Saskatchewan AB. 76 pp.

Cody, M.L. 1986. Diversity, rarity and conservation in Mediterranean climate regimes. In: Conservation biology - the science of scarcity and diversity. M.E. Soule (ed.). Sinauer Associates. Sunderland MS. Pp. 112-152.

Cool, N. 2003. Personal communication. Park Warden/Wildlife Biologist, Elk Island National Park. Fort Saskatchewan AB.

Cool, N. 1999. Elk Island National Park ecosystem management plan. Unpubl. Rep. Fort Saskatchewan AB. 47 pp.

Cooke, A.S. and L. Farrell. 2001. Impact of muntjac deer *Muntiacus reevesi* at Monks Wood National Nature Reserve, Cambridgeshire, Eastern England. Forestry 74:241-250.

Cooke, A.S. and K.H. Lakhani. 1996. Damage to coppice regrowth by muntjac deer *Muntiacus reevesi* and protection with electric fencing. Biol. Conserv. 75:231-238.

Detling, J.K. 1998. Mammalian herbivores: ecosystem-level effects in two grassland national parks. Wildl. Soc. Bull. 26:438-448.

DePuit, E.J. 1986. Western revegetation in perspective: past progress, present status and future needs. In: Proc. High Altitude Revegetation Workshop No. 7. CWRRI Information Ser. 58. M.A. Schuster and R.H. Zuck (eds.). Pp. 6-34.

Dobson, A.P., A.D. Bradshaw and A.J.M. Baker. 1997. Hopes for the future: restoration ecology and conservation biology. Science 277:515-522.

Ducks Unlimited Canada. 1995. Revegetating with native grasses. Native Plant Materials Committee. Stonewall MB. 133 pp.

Ehrenfeld, J.G. 2000. Defining the limits of restoration: the need for realistic goals. Restor. Ecol. 8:2-9.

Engstrom, R.T., S. Gilbert, M.L. Hunter, Jr., D. Merriwether, G.J. Nowacki and P. Spencer. 1999. Practical applications of disturbance ecology to natural resource management. In R.C. Szaro, N.C. Johnson, W.T. Sexton and A.J. Malk (eds.). Ecological stewardship; a common reference for ecosystem management. Vol. II. Elsevier Science, Ltd. Oxford UK. Pp. 313-330.

Flather, C.H. and T.W. Hoekstra. 1989. An analysis of the wildlife and the fish situation in the United States: 1989-2040. In: General Technical Report. RM-178: Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. Fort Collins CO. Pp. 1-146.

Franco, M. and J.L. Harper. 1988. Competition and the formation of spatial pattern in spacing gradients: an example using *Kochia scoparia*. J. Ecol. 76:959-974.

Frank, D.A. and S.J. McNaughton. 1993. Evidence for the promotion of aboveground grassland production by native large herbivores in Yellowstone National Park. Oecologia. 96:157-161.

George, T.L. and S. Zack. 2001. Spatial and temporal considerations in restoring habitat for wildlife. Restor. Ecol. 9(3)272-279.

Gerling H.S., M.G. Willoughby, A. Schoepf, K.E. Tannas and C.A. Tannas. 1996. A guide to using native plants on disturbed lands. Alberta Agriculture, Food and Rural Development, Alberta Environmental Protection. Edmonton AB. 247 pp.

Grime, J.P., J.M.L. Mackey, S.H. Hillier and D.J. Read. 1987. Floristic diversity in a model system using experimental microcosms. Nature 328:420-422.

Grootjans, A.P., H. Everts, K. Bruin and L. Fresco. 2001. Restoration of wet dune slacks on the Dutch Wadden sea islands: recolonization after large scale sod cutting. Restor. Ecol. 9:137-146.

Hardy BBT Limited. 1989. Manual of plant species suitability for reclamation in Alberta 2nd Edition. Alberta Land Conservation and Reclamation Council Report No. RRTAC 89-4. Edmonton AB. 436 pp.

Hartnett, D.C. 1989. Density and growth stage dependent responses to defoliation in two rhizomatous grasses. Oecologia 80:414-420.

Higgs, E.S. 1997. What is good ecological restoration? Conserv. Biol. 11:338-348.

Higgs, E.S. 1994. Expanding the scope of restoration ecology. Restor. Ecol. 2:137-147.

Hobbs, R.J. and D.A. Norton. 1996. Towards a conceptual framework for restoration ecology. Restor. Ecol. 4:93-110.

Hobbs, N.T. 1996. Modification of ecosystems by ungulates. J. Wildl. Manage. 60:695-712.

Hobbs, N.T., D.S. Schimel, C.E. Owensby and D.J. Ojima. 1991. Fire and grazing in the tallgrass prairie: contingent effects on nitrogen budgets. Ecology 72:1374-1382.

Hudson, R.J. and M.T. Nietfeld. 1985. Effect of forage depletion on the feeding rate of wapiti. J. Range Manage. 38:80-82.

Janzen, D.H. 1984. Dispersal of small seed by big herbivores: foliage is the fruit. Am.

Nat. 123:338-353.
Jordan, W.R., M.E. Gilpin and J.D. Aber. 1987. Restoration ecology: a synthetic approach to ecological research. Cambridge University. New York NY. 342 pp.
Kay, C.E. 1997. Viewpoint: ungulate herbivory, willows and political ecology in Yellowstone. J Range Manage. 50:139-145.
Keck, W.M. 1972. Great Basin station- sixty years of progress in range and watershed research. Res. Pap. INT-118.: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. Ogden UT. Pp. 1-48.
Kentula, M.E. 1997. A step towards a landscape approach in riparian restoration. Restor. Ecol. 5:2-3.
Kitteridge, D.B. and M.J. Kelty. 1992. The use of tree shelters with Northern Red Oak natural regeneration in southern New England. N. J. Appl. For. 9:141-145.
Lamb, T.L. 1998. A study of plant community structure and a reclamation evaluation of disturbed subalpine sites in Glacier National Park, B.C. M.Sc. thesis. University of Alberta. Edmonton AB. 89 pp.
Majerus, M. 1997. Restoration of disturbances in Yellowstone and Glacier National Parks. J. Soil Water Conserv. 232-236.
McArthur, E.D. 1988. New plant development in range management. In: Vegetation science applications to rangeland analysis and management. P.T. Tueller (ed.) Kluwer Academic Publishers. Boston MA. Pp. 81-112.
Meeuwig, R. O. 1960. Watersheds A and B, A study of surface runoff and erosion in the subalpine zone of central Utah. J. For. 58:556-560.
Morgan, J.P. and D.R. Collicutt. 1995. Restoring Canada's native prairie: a practical manual. Prairie Habitats. Argyle MB. 84 pp.
Morrison, M.L. 2001. Introduction: concepts of wildlife and wildlife habitat for ecological restoration. Restor. Ecol. 9:251-252.
Naeth, M.A. 1999. Renewable Resources 475. Vegetative reclamation, Revegetation and Restoration. Class lecture notes. Dept. Renewable Resources, University of Alberta. Edmonton AB.
Ocumpaugh, W.R., S. Archer and J.W. Stuth. 1996. Switchgrass recruitment from broadcast seed vs. seed fed to cattle. J Range Manage. 49:368-371.
Osko, T.J., R.T. Hardin and B.A. Young. 1993. Research Observation: Chemical repellents to reduce grazing intensity on reclaimed sites. J. Range Manage. 46:383-386.
Parks Canada. 1994. Guiding principles and operational policies. Minister of Supply and Services Canada. Ottawa ON. 125 pp.
Parks Canada Agency. 2001. Western Canada restoration ecology guiding principles- Draft. Parks Canada/University of Alberta Workshop. Terrestrial Ecological Restoration Guiding Principles Working Paper. Unpubl. Rep. 5 pp.
Parks Canada Agency. 2000. Unimpaired for future generations? Protecting ecological integrity with Canada's National Parks. Vol. I. A call to action. Vol. II. Setting a new direction for Canada's National Parks. Report of the panel on the ecological integrity of Canada's National Parks. Ottawa ON. 147 pp.
Pelech, W.E. 1997. Performance of selected native and introduced plant species under mowing and herbicide management during the establishment period. M.Sc. thesis. University of Alberta. Edmonton AB.

Pfadenhauer, J. 2001. Some remarks on the socio-cultural background of restoration ecology. Restor. Ecol. 9:220-229.

Pfadenhauer, J. and A.P. Grootjans. 1999. Wetland restoration in Central Europe: aims and methods. Appl. Veg. Sci. 2:95-106.

Pitchford, C. 1999. Season of seeding, mowing and seed mix richness for native plant community development in the Aspen Parkland. M.Sc. thesis. University of Alberta. Edmonton AB. 230 pp.

Polley, H.W. and S.L. Collins. 1984. Relationships of vegetation and environment in buffalo wallows. Am. Midl. Nat. 112:178-186.

Richards, R.T., J.C. Chambers and C. Ross. 1998. Use of native plants on federal lands: policy and practice. J. Range Manage. 55:625-632.

Roundy, B.A., N.L. Shaw and D.T. Booth. 1997. Proc.: using seeds of native species on rangelands. Society for Range Management 50th Annual Meeting. Rapid City S.D. Pp. 1-30.

Sinclair, A.R.E., S.A.R. Mduma and P. Arcese. 2002. Protected areas as biodiversity benchmarks for human impact: agriculture and Serengeti avifauna. Proc. of the Royal Society of London series B-Biological Sciences. 269:2401-2405.

Sinclair, A.R.E. 1998. Natural regulation of ecosystems in protected areas as ecological baselines. Wildl. Soc. Bull. 26:399-409.

Singer, F.J. 1995. Effects of grazing by ungulates on upland bunchgrass communities of northern winter range of Yellowstone National Park. Northwest Sci. 69:191-203.

Smallwood, K.S. 2001. Linking habitat restoration to meaningful units of animal demography. Restor. Ecol. 9:253-261.

Society for Ecological Restoration International Science and Policy Working Group. 2002. The SER primer on ecological restoration. www.ser.org.

Society for Ecological Restoration. 1990. Minutes of the annual meeting of the Board of Directors. Society for Ecological Restoration. Madison WI.

Stoddart, L.A., A.D. Smith and T.W. Box. 1975. Range management. McGraw-Hill. New York N.Y. 532 pp.

Sutton, R.K. 1975. Why native plants aren't used anymore. J. Soil Water Conserv. 30:240-242.

Takyi, S. K. and R.M. Islam. 1984. Performance of native grasses and cultivated legumes and grasses and disturbances in the eastern slopes of the Rockies. Alberta Energy and Natural Resources, Forest Service. Edmonton AB. Pp. 1-30.

Taylor, R.A.J. and L.R. Taylor. 1979. A behavioural model for the evolution of spatial dynamics. In R.M. Anderson, B.D. Turner and L.R. Taylor (eds.). Population Dynamics. Blackwell Scientific Publications. Oxford UK. Pp. 1-28.

Trottier, G.C. 1992. Conservation of Canadian prairie grasslands: a landowners guide. Environment Canada, World Wildlife Fund and North American Waterfowl Management Plan. 92 pp.

Vallentine, J.F. 1989. Range development and improvements. Academic Press. San Diego CA. 524 pp.

Van der Valk, A.G. 1975. Floristic composition and structure of fen communities in northwest Iowa. Proc. of the Iowa Academy of Science 82:113-118.

Van Diggelen, R., A.P. Grootjans and J.A. Harris. 2001. Ecological restoration: State of the art or state of the science? Restor. Ecol. 9:115-118.

Van Wieren, S.E. 1995. The potential role of large herbivores in nature conservation and extensive land use in Europe. Biol. J. Linnean Soc. 56:11-23.

Vinton, M.A., D.C. Hartnett, E.J. Finck and J.M. Briggs. 1993. Interactive effects of fire, bison *Bison bison* grazing and plant community composition in tallgrass prairie. Am. Midl. Nat. 129:10-18.

Vinton, M.A. and D.C. Hartnett. 1992. Effects of bison grazing on *Andropogon gerardi* and *Panicum virgatum* in burned and unburned tallgrass prairie. Oecologia 90:374-382.

Vinton, M.A. 1990. Bison grazing patterns and plant responses on Kansas tallgrass prairie. M.Sc. thesis, Kansas State University. Manhattan KS. 57 pp.

Vogel, W.G. 1987. A manual for training reclamation inspectors in the fundamentals of soils and revegetation. U.S. Dept. of Agric. Berea KY. 178 pp.

Young, T.P. 2000. Restoration ecology and conservation biology. Biol. Conserv. 92:73-83.

CHAPTER 2. UNGULATE HABITAT RESTORATION IN ELK ISLAND NATIONAL PARK

2.1 INTRODUCTION

Sustaining the integrity of park ecosystems has become a key issue in National Park management. Parks seldom contain complete and unaltered ecosystems due to degradation from invasion of non-native species, climate change and human activities. National Parks are mandated to make every effort to maintain natural processes within their boundaries; however, techniques restricted by national park guidelines are frequently used outside of their jurisdiction. Examples of this mandate are that seeding non-native species on disturbed areas inside the park is not permitted and that all efforts must be made to prevent the introduction of non-native plants into the national parks.

Park staff can use ecosystem management as a tool to protect ecosystems and direct recovery from stressful human activities. When ecosystem functions are altered by human activities, and natural processes will not likely achieve reclamation objectives, intervention may be justified to restore ecological integrity. Park managers must use scientific research and monitor disturbed areas to ensure natural processes are restored.

Elk Island's current five-year management plan stipulates that all non-natural areas situated inside its boundaries must be restored. Disturbed areas are usually reseeded with native prairie grasses that are indigenous to the park. On occasion, the park has utilized trees, clumps of trees, shrubs and clumps of shrubs to restore disturbed forest and shrubland. Transplants of these various types of vegetation are removed from adjacent areas and positioned onto disturbed sites. This guarantees national park policy is adhered to by conservation biologists working in Elk Island National Park (EINP) (Chapman 2003).

This research involved revegetating recently disturbed historic landfill sites for an end land use of habitat restoration for ungulates. The sites will serve as references for future revegetation projects located in the park. If the native grass mix flourishes, its components could be used on other disturbed areas which may arise from park activities.

Stress on revegetated sites occurs when ungulates trample newly seeded sites, trample emerging plants, graze emerging plants, bed in newly seeded areas and, in the case of bison, create wallows. Each of these ungulate activities has the potential to hinder or prevent a successful revegetation project. When ungulates stress emerging vegetation, revegetation projects may require fencing prior to seeding and at least during the early stages of plant community development. Long term success of a restoration project may then hinge on post-establishment grazing management (Archer and Pyke 1991).

Restoration success may be affected by the type of seeding method used. Although broadcast seeding may create a more natural looking plant community, the better soil-seed contact afforded with drill seeding may prove more successful. Restoration activities may occur in areas not accessible by drill or too small to drill seed and therefore broadcast seeding may be required. Some restoration species may have limited available seed and it may be costly; thus broadcast seeding, which requires a higher planting density than drill seeding, may be less economical.

Future restoration projects in national parks require an adaptive management approach that draws from the best available information, assesses management alternatives and takes action in the context of scientific experiments (Boyce 1998, Porter and Underwood 1999, Wright 1999). Thus the seeding method and fencing treatments were required to determine factors that would potentially enhance restoration success.

2.2 OBJECTIVES AND HYPOTHESES

2.2.1 RESEARCH OBJECTIVES

The overall research objectives were to compare fencing and seeding treatments in the context of reclamation/restoration and to assess the extent of the use of the newly revegetated sites by native ungulates. Specific objectives were to:

- Compare drill seeding and broadcast seeding of native plant species as they impact select vegetation parameters.
- Determine if fencing had an impact on select vegetation parameters.
- Monitor wildlife use of the research treatments.

- Evaluate the seed mix for restoration potential.

2.2.2 Hypotheses

The experimental design tested the following null hypotheses in the context of plant and plant community parameters: drill seeding is the same as broadcast seeding; fencing is the same as not fencing.

Fencing newly seeded disturbed areas in the park was deemed necessary, due to the overabundance of wildlife in Elk Island. Wildlife interactions inhibit plant emergence and growth, postpone reductions of bare ground and consequently produce less biomass in revegetated areas. The experimental design tested the following alternative hypothesis in the context of plant and plant community parameters: revegetation success in unfenced areas will not be as successful as those conducted in fenced settings.

Drill seeding provides better soil seed contact which minimizes wind erosion at newly seeded sites. Emerging drill seeded species rapidly displace bare ground and compete against non-native species naturally occurring in the seed bank. Establishment of native species in the seed mix will provide additional biomass for ungulates in the park. Broadcast seeding does not provide as good soil seed contact as drill seeding and seeds become highly susceptible to wind and water erosion as well as predation. Loss of seeds will decrease the number of seeded species, increase non-seeded species and offer a reduced amount of biomass for ungulates.

2.3 SITE DESCRIPTION

2.3.1 Site History

The two main disturbed areas that were studied served as landfills for EINP. The Site 1 landfill was created in the 1930s and decommissioned in the early 1970s (Chapman 2003). The Site 2 landfill was created at the time of the park establishment (1900s or early 1910s) and was decommissioned in the 1960s (Chapman 2003). Refuse in Site 1 was initially covered with soil and untouched until summer 1997. Most refuse at Site 2 was covered with soil although some surface dumping occurred. After decommissioning, the Site 2 landfill was untouched until summer 1997. After refuse removal by park staff

the sites were denuded of vegetation. The landfilled material was removed and the heavy equipment operator used the bucket on the front end loader to contour the sub soil at the sites to predisturbance levels. Topsoil was not replaced at either site and compaction resulted from the use of heavy equipment.

When the landfilled sites were cleaned out, soil testing was done by O'Connor Associates to ensure there were no contaminants in the landfills. All of the analyses indicated the sites met the required government criteria (Tables A1 through A3).

2.3.2 Location

EINP is located in a transitional zone between the Aspen Parkland and the Lower Boreal Mixedwood ecoregions in central Alberta (Strong and Leggat 1992, Bork et al. 1997). The park is approximately 45 km east of Edmonton on Highway 16 and the highway divides the park into northern and southern sections (Figure A1). The northern portion is divided by the Parkway, which runs north from Highway 16 to the boundary of the park, abutting Township Road 550.

Sites for this research are located in the northern portion of the park and are approximately two km apart (Figure A2). Legal land descriptions for Site 1 and 2 are NW-13-54-20-W4M and SW-14-54-20-W4M, respectively. Both research sites became disturbed in summer 1997 when Park staff reclaimed landfills that had some surface dumping as well as refuse buried beneath a shallow layer of topsoil. They were decommissioned in August 1997.

Site 1 is located 13.9 km from the park entrance at Highway 16 and positioned approximately 300 m east of the parkway, directly across from the golf course equipment shop. The entire site encompasses approximately 480 m^2 with a fenced area occupying approximately 240 m^2. The fenced plots are situated approximately 1.5 m directly west of the unfenced treatments. The area of the research plots situated inside and outside the fenced area is 192 m^2. The combined area for both main plots is 384 m^2. Trees border the east, west and south sides of the site and an open meadow is located north of the site.

Site 2 is located 10.8 km from the park entrance at Highway 16 approximately 500 m north of the parkway. The entrance to the site is approximately 250 m east of the Administration Road turn off and follows a decommissioned road that is about 600 m in

length. The entire site encompasses approximately 2215 m^2 with a fenced portion about 470 m^2 in size. The research plots situated inside and outside the fenced area are 420 m^2, for a combined main plot area totalling 840 m^2. The unfenced treatment is partially bordered on the north and west by trees while grasslands and wetlands border the remaining edges. The fenced treatment is northeast of the unfenced plots and has trees to the west and wetlands to the north and east.

Two other disturbed sites were studied. Site 3 was located 350 m west of the parkway across from the Information Center. The area was disturbed in summer 1997 when Park staff removed a footing from an old shop. Site 4 was situated 500 m south of Highway 16 near Goose Lake. The area was disturbed in spring 1998 when Park staff searched ground squirrel nests. Data were collected in 1998 and 1999. In spring 2000, park staff conducted a controlled burn that encompassed Site 3. An assessment at Site 4 at the end of the 2000 growing season revealed the site was overrun by agrimony (*Agrimonia striata* Michx.). Since no seeded species were observed inside the plots after the third growing season, success of seeded species could not be assessed. Due to extenuating circumstances at Sites 3 and 4, statistical analyses focused only on Sites 1 and 2.

2.3.3 Climate

The park falls in a transition zone between a grassland and a boreal forest type climate (Strachan et al. 1986). Park records indicate that January is the coldest month with a mean temperature of -13.3 °C and July is the warmest month with a mean temperature of 16.6 °C (Environment Canada 2003a). Total annual precipitation in EINP ranges from 400 to 500 mm with 78% occurring as summer precipitation and the balance occurring as snowfall (Kerr et al. 1993). Mild winters arising from Chinook winds are uncommon and frost may occur 245 to 275 days of the year (Kerr et al. 1993).

2.3.4 Topography

EINP is situated at the north end of the Cooking Lake Moraine, which is approximately 70 m higher in elevation than the surrounding terrain (Strachan et al. 1986). Present topography was created from land forming processes brought about by the disintegration of a stagnant ice sheet left behind from deglaciation. The five types of

surficial deposits in the park are till, glaciofluvial, glaciolacustrine, organic and lacustrine (Jennings 1983). Both of the research sites are located on glaciolacustrine landforms dominated by undulating to hummocky glacial till deposits (Kerr et al. 1993).

The treatments at Site 1 have a south-facing aspect with approximately 9 to 11% slope in the fenced treatments and 10 to 12% slope on the unfenced treatments. Fenced treatments at Site 2 are on a north-facing aspect with a 10 to 12% slope and unfenced treatments have a south-facing aspect with an 8 to 10% slope.

2.3.5 Hydrology

Surface waters cover approximately 11% of the total area of the park (Strachan et al. 1986). Topography supports an abundance of water-filled depressions most of which are small in size. A small ephemeral creek flows from the north to the east of the fenced treatment at Site 2; however, it is poorly developed and only flows during snowmelt runoff. Groundwater flow patterns for each site are towards the boundaries of the Park.

2.3.6 Geology And Soils

Elk Island is located in the Alberta Plains physiographic area of the Interior Plains region (Bostock 1970). Bedrock underlying surface deposits are from the late Cretaceous period making up the Edmonton formation which ranges from 0 to 122 m and consists of fine grained calcareous and bentonitic sandstones; sandy, bentonitic and carbonaceous shales; mudstone; volcanic tuff, bentonite, ironstone bands and coal seams (Douglas 1970). AGRASID (1998) classifies the area as the Islet Upland with Gray and Dark Gray Luvisols developed on medium textured till as the dominant soil groups and Gleysols and Chernozems as minor soil groups.

Site 1 primary soils belong to unit (NC-COA) 1 (Crown 1977). The dominant soil is an Eluviated Eutric Brunisol and the subdominant soil is an Orthic Gray Luvisol. Parent material is glaciolacustrine sediments and glacial till. Two secondary soils, units (COA-NC) 1 and (COA-HDY) 1 are medium to moderately fine textured glacial till with a veneer of coarse textured glaciofluvial sediments. Soils to 20 cm are loam or sandy loam and greater depths are clay loam in (NC-COA) 1, loamy sand in (COA-NC) 1 and sandy loam underlain by clay loam and loam in (COA-HDY) 1. Soils are well drained in (COA-HDY) 1, well to rapidly drained in (COA-NC) 1 and rapidly to well drained in

(NC-COA) 1. Soils in (COA-NC) 1 are highly susceptible to erosion while other units are moderately susceptible. Slopes are 6 to 15% in (COA-HDY) 1 and 10 to 25% in (COA-NC) 1 and (NC-COA) 1. Soil pH varies from 4.5 in A horizons of (COA-NC) 1 and (NC-COA) 1 to 7.5 in C horizons of (COA-HDY) 1. Electrical conductivity is lower than 1 denoting non-saline soils. Soils have a 5 to 8 cm LF layer overlaying the A horizon. Water erosion is a moderate hazard when vegetation is removed (Crown 1977).

The Site 2 primary soil belongs to soil units (MIQ-COA) 2 and (COA-HDY) 1. Orthic Gray Luvisol is dominant and Dark Gray Luvisol is subdominant. Soil unit (MIQ-COA) 2 is medium to moderately fine textured glacial till with a veneer of stonefree, fine textured, glaciolacustrine sediments of variable thickness (Crown 1977). Soil depths up to 12 cm are sandy loam and greater depths are clay loam to clay. Soils are moderate to well drained, moderate to highly susceptible to erosion with 6 to 15% slopes. Soil pH in unit (MIQ-COA) 2 varies from 5.5 in the A horizons to 7.8 in the C horizon. Electrical conductivity is lower than 1 denoting soils are non-saline. This unit has an LF layer approximately 4 cm in depth overlaying the A horizon. The water erosion hazard is high for slopes greater than 9% when vegetation is removed (Crown 1977).

2.3.7 VEGETATION

The Aspen Parkland Ecoregion is dominated by trembling aspen (*Populus tremuloides* Michx.), willow (*Salix* L. spp.) and grasses (*Agropyron* Gaertn. spp., *Bromus* L. spp., *Stipa* L. spp., *Poa* L. spp. and *Festuca* L. spp.). White spruce (*Picea glauca*) is the dominant conifer in the area. Some species found on adjacent land bordering the sites are balsam poplar (*Populus balsamifera* L.), trembling aspen, saskatoon (*Amelanchier alnifolia* Nutt.), streambank wheatgrass (*Agropyron riparium* Scribn. & Smith) and western wheatgrass (*Agropyron smithii* Rydb.) (Strachan et al. 1986). Detailed lists of plant species identified on the sites are in Tables A4 through A7.

2.3.8 WILDLIFE

EINP supports a wide diversity and abundance of wildlife. Presently, 44 species of mammals, 240 species of birds, 6 species of reptiles and amphibians and 3 species of fish have been identified in the Park (Chapman 1991). The park has a large density of ungulates: elk (*Cervus elaphus*), moose (*Alces alces*), plains bison (*Bos bison bison*),

mule deer (*Odocoileus hemionus*) and white-tailed deer (*Odocoileus virginianus*). Wood bison (*Bos bison athabascae*) are confined to the southern section of the park and thus could not affect revegetation. Ungulate densities are five times greater than controlled wildlife management areas in Alberta with approximately 10 to 12 ungulates km^{-2} inside the park and 2 to 5 ungulates km^{-2} in wildlife systems in the province (Cool 2003).

2.4 MATERIALS AND METHODS

2.4.1 TREATMENTS AND EXPERIMENTAL DESIGN

The total area of each block (site) differed due to disturbance size. Main plots (fencing treatments) at Sites 1 and 2 were 192 and 420 m^2, respectively. The largest possible subplots were staked for both fenced and unfenced treatments. Unfenced treatments were situated east of fenced treatments in Site 1 and southeast of fenced treatments in Site 2. Fenced treatments comprised four randomized subplots, each 6 x 8 m at Site 1 and 10 x 10.5 m at Site 2. Two of the four subplots in each fencing treatment were drill seeded and the other two were broadcast seeded. Seeding treatments were randomly selected fixed factors applied to each experimental unit. Each drill seeded experimental unit (subplot) at Site 1 required 3 seed drill passes for a total of 24 rows, each 8 m long. At Site 2, each drill seeded experimental unit required 5 passes for a total of 40 rows, each 10.5 m long. Figure A3 provides a layout of the main plots and seeding treatments for the study sites.

The design structure was a split plot with the main plots in a randomized complete block design (RCB) (Figure A3). The design can be further classified as a 2 x 2 factorial experiment. There are two factors, fencing and seeding method and each factor has two levels. The fence factor levels are fencing and no fencing and the seeding factor levels are drill seeding and broadcast seeding. Split plot designs are frequently used for factorial experiments and can incorporate a randomized complete block design (Steel et al. 1997).

This is a split plot experiment because there are two different sized experimental units in each block. Each block is divided into two main plots (experimental units) that

receive one of two types of fencing. These experimental units are divided into subplots that receive one of two types of seeding.

It is a RCB design because blocks were randomly selected. Each site is a block since it contains all possible treatment combinations of the experiment. A RCB design was chosen because the power of the experiment could be increased if the experimental units could be grouped into blocks (Blenis 1999). Blocking does not become a treatment because it is part of the design structure and it removes an extraneous amount of variation (Steel et al. 1997). The variation among units within a block is less than that among units in different blocks and the resulting precision of the experiment is increased as a result of error control (Steel et al. 1997).

Interactions between fencing and seeding treatments gave a total of four treatment combinations. The combinations were: fenced and drill seeded, fenced and broadcast seeded, unfenced and drill seeded and unfenced and broadcast seeded. Each of the four treatment interactions has been replicated at each site. There are eight experimental units in each block, for a total of 16 experimental units. The statistical linear models can found in Tables A8 and A10 and Tables A9 and A11 contain the ANOVAs.

2.4.2 Seed Mix

Both disturbed sites at Elk Island were revegetated with the same native species seed mix (Table 1). Native species were selected because of their ability to grow in a nutrient poor environment and park guidelines stipulated only native species could be used in reclamation. The seed mix comprised early and late growing perennial species. A description of individual plant suitability criteria can be found in Table A12.

Each species of seed was weighed to the nearest gram then total seeds per gram were counted. This process was completed three times and an average was calculated for number of seeds per gram for each species. The seeding rate for drill seeded treatments was approximately 300 pure live seeds (PLS) m^{-2} (determined by 20/20 Labs) and broadcast seeded treatments were seeded at 600 PLS m^{-2} to yield approximately 300 plants m^{-2} in each treatment. The gram average for each species was multiplied by the seeding rate, the subplot area, % PLS and finally by one sixth (six species were used

evenly in the seed mix). This process calculated the weight of seed required of each species in each subplot as gram plot $^{-1}$.

Drill seeding calculations required one more step to determine seed weight required for each row. The gram plot^{-1} value was multiplied by the number of rows per subplot and expressed as gram row^{-1}. Seeds for each individual row were packaged into envelopes to facilitate loading of the drill seeder cones. Broadcast seeding totals for each subplot were calculated by adding the gram plot^{-1} values of each species, then all species were deposited into an envelope and shaken to mix the seed. Chick starter was used as a carrier in both broadcast and drill seeding to facilitate movement of the seeds through the cones on the drill seeder and through the openings on the broadcast seeder.

2.4.3 Seed Bed Preparation and Seeding

Both sites were sprayed with glyphosate on June 10, 1998 according to manufacturer directions and then rototilled on June 17, 1998. Due to park policy, topsoil was not brought into the sites prior to seeding to prevent potential undesirable species in the seed bank from establishing inside the plots. The amount of refuse, which was at times clearly visible on top of the soil, did not permit the contractor to stockpile the topsoil layer (Kratko 1998). Since the topsoil could not be saved, the native plant species mix was seeded directly onto disturbed subsoil.

The seed mixes were broadcast and drill seeded on June 19 and 20, 1998. The drill seeder covered a 2 m swath for each pass consisting of eight rows. The drill seeder was equipped with eight cones that placed the rows at 20 cm intervals and seeded them to a preset depth of 2.5 cm. The broadcast seeding was completed with a handheld broadcast seeder which spread the seed evenly on each of the four subplots. After seeding the broadcast seeded subplots were raked to ensure good seed-soil contact and to prevent water and wind erosion. Broadcast seeding took place when there was relatively no wind to minimize seed drifting. A 4 x 8 sheet of plywood was set up between the plots to ensure seeds being broadcasted were not dispersed onto the drill seeded portions of the sites or outside of the broadcast seeded plots.

2.4.4 FENCING

Fences were constructed at each site in spring 1998 to prevent ungulate interactions (grazing, trampling, wallowing, etc.) from occurring in half of the main plots. Fences were constructed with treated, 20.3 cm diameter, 3.7 m length poles which were pounded into the ground with a backhoe supplied from Parks Canada. Game fence (15.2 x 15.2 cm squares) similar to that used for park boundaries was then fastened to the poles with horseshoe nails. A 3 m gate was constructed in the fence to permit access for equipment. The completed fence was 2.5 m high.

2.4.5 SAMPLING AND MEASUREMENTS

Assessments were conducted in 1998, 1999 and 2000. Soil was sampled once in 1999 and vegetation biomass was sampled once in 2000. Plant species richness and wildlife utilization were assessed in the final two growing seasons (1999 and 2000) and ground cover was observed over all three years.

2.4.5.1 Meteorological

Air temperature and precipitation data were collected in 1998, 1999 and 2000 from the meteorological station at the EINP warden station less than two kilometers away. Data collected from Elk Island were compiled by Environment Canada. Average monthly temperature, total monthly and total yearly precipitation from the 1998 to 2000 growing seasons was compared to 1971 to 2000 long-term averages.

2.4.5.2 Soil

Soil was sampled at Site 1 on October 27th, 1999 and at Site 2 on November 4th and 10th, 1999. Stratified random samples were required at both sites due to the sparse areas where the seeded mixture had emerged. Each core was extracted with a Dutch auger and placed in the appropriately marked plastic bag. Both sites had three random samples which were each comprised of three subsamples for fenced and unfenced treatments making a total of 12 samples and 36 subsamples. Samples 1 to 6 were taken at Site 1 and samples 7 to 12 were taken from Site 2. Four depths were assessed in each subsample: 0 to 10 cm, 10 to 20 cm, 20 to 60 cm and 60 to 100 cm. Cores from each subsample were placed in the same corresponding depth bag, for a total of four bags per

sample. The total number of soil samples (12) and the four depths associated made a sample total of 48.

Samples were air dried, crushed and ground to pass a 2 mm sieve at the Ellerslie Research Station. They were then taken to the University of Alberta and samples 1, 3, 6, 8, 11 and 12 were randomly selected for analyses. Samples 1, 3 and 6 were extracted from Site 1 and samples 8, 11 and 12 were taken from Site 2. Samples 1, 3, 11 and 12 were from unfenced treatments and samples 6 and 8 were taken from fenced treatments. Particle size analyses (PSA), hydrogen ion concentration (pH), electrical conductivity (EC) and total organic carbon were determined for each sample.

PSA was completed by the hydrometer method which determined percentages of clay, silt and sand (McKeague 1978, Sheldrick and Wang 1993). The Canadian System of Soil Classification (1978) textural triangle was used to establish the soil textural class for each sample. Chemical property analyses were conducted at the University of Alberta Soil Lab and compared to the international standard and Ellerslie fallow.

2.4.5.3 Vegetation

Ground cover was assessed at the end of each growing season (August) in 1998, 1999 and 2000. Each subplot was randomly assessed by a different technician in each of the three growing seasons to ensure each experimental unit was assessed without bias. In each subplot, at each site, five 0.1 m^2 randomly located quadrats were assessed for bare ground, litter, live vegetation, moss, dung and rocks and based out of 100. Species area curves were created to confirm that the number of samples observed would contain the majority of species in each subplot. Live vegetation was divided into grasses, forbs, weeds, shrubs and trees. In 1999 and 2000, live vegetation was identified to species.

After live vegetation from each quadrat was identified to species, it was further determined how many individuals of each species were present (density) and then the total area covered by each species was estimated as a percentage of live vegetation (biovolume). These parameters were further divided into seeded and non-seeded species. Non-seeded species consisted of any species not in the seed mix.

Two 1.5 m diameter range cages were randomly placed in each of the four unfenced subplots on May 5, 2000 to assess potential biomass available for ungulates

during the final growing season. In summer 2000, a 0.5 m^2 quadrat was placed in the center of the range cages and vegetation was clipped with an electric clipper to ground level to simulate grazing. This process was replicated in fenced and unfenced treatments for a total of 24 clippings from each site (8 from the fenced, 8 from unfenced and 8 from range cages). Collected vegetation was then kiln dried and weighed.

2.4.5.4 Wildlife Utilization

Each site was monitored for wildlife use and detailed assessments were conducted in the final two growing seasons. Each subplot was divided into m^2 grids and each m^2 was assessed for wildlife activities. Wildlife activities were categorized as bedding, grazing, trampling, wallowing and scat (feces). Visual verification, pellets and tracks were used to identify ungulates that utilized the sites. Ungulates identified through these techniques were elk (*Cervus elaphus*), moose (*Alces alces*), plains bison (*Bos bison bison*) and deer spp. (*Odocoileus hemionus* and *Odocoileus virginianus*).

2.4.6 Data Analyses

Data from ground cover, biomass, plant density and plant biovolume were analyzed using analysis of variance (ANOVA) in SAS. When using ANOVA to determine tests of significance basic assumptions need to be met. Treatments must be additive and experimental errors have to be random, independently and normally distributed about a zero mean with a common variance (Steel et al. 1997). Steel et al. (1997) indicated that for types of biological data, experience indicates the usual disturbances resulting from failure of the data to fulfil the above requirements are unimportant. Treatments in this study were assigned to the experimental units at random and the order of observations was determined at random. The effect of the randomization process was to make the errors independent of each other (Steel et al. 1997).

The PROC GLM procedure of SAS was used to ensure that the basic assumptions were adhered to prior to analyzing the data. The PROC GLM procedure was conducted on the absolute values of the residuals to determine homogeneity. The former analysis is referred to as the Levene's test and it is a robust test that is less sensitive to departures from normality but sensitive to heterogeneous variances (Milliken and Johnson 1984).

The Shapiro-Wilk Test was used to check for normality because it is a good test for sample sizes up to 50. The underlying assumption of homogeneity of variance could not be assumed since there were only single degrees of freedom.

The analyses were run using the PROC MIXED procedure of SAS due to the split plot design. The MIXED procedure (Steel et al. 1997) in SAS was used because unlike PROC GLM, calculated values of the main plot error term would not assume a negative variance for the main plot experimental error and it could be used to model heterogenous variances in situations where the assumption of homogeneity of variance was violated (Blenis 2000). Satterthwaite's approximation was used to calculate denominator degrees of freedom. A p-value of 0.10 was used instead 0.05 to prevent committing a Type II error due to the low power of the test. Steel et al. (1997) state that p-value levels of 5 and 1 % are adequate for agricultural tests but in the case of small sized experiments, the null hypothesis will not likely be rejected at the lower significance levels and therefore a significance level of 10 % should be used to prevent a Type II error.

Separate statistical analyses for ground cover (bare ground, litter, live vegetation), and species composition (seeded, non-seeded species) were conducted on all three years of growing season data. Although the trend for vegetation to increase over the three growing seasons was not at question, whether these changes occurred in each growing season was of interest. Of specific interest was which treatment combinations produced significant differences among ground cover and species composition parameters after the third growing season.

Statistical analysis was completed on biomass data from range cages, fenced and unfenced treatments at the end of each growing season. Analyses were conducted to determine if significant differences occurred between fenced and unfenced treatments and between range cages and unfenced treatments. Range cage and the fenced treatment data were not compared since the range cages were placed on unfenced treatments after two years of grazing and the fenced treatments had not experienced any grazing during the same time period. The trend for biomass to increase over the three growing seasons was expected but of specific interest was biomass differences from the four treatment combinations at the end of the third growing season.

2.5 RESULTS AND DISCUSSION

All the dependent variables in each of the three growing seasons were normally distributed but with heterogeneous variances. Thus no transformations of any of the data were required. None of the random effects (site and site by fence interaction) of the model were significant. No statistically significant differences were found for any of the treatment interactions (seeding by fencing combinations) for all dependent variables in all three years. Thus the statistical analyses and discussions focused only on the main effects of seeding method and fencing.

2.5.1 METEOROLOGICAL

Data compiled from Environment Canada (2003b) showed that average monthly temperatures tended to be slightly higher than the 1971 to 2000 normals over the study period while precipitation was lower (Table 2). Precipitation differences from normals occurred mainly in the growing seasons. A meteorological point of interest occurred the first growing season with approximately 30 mm more precipitation than the normals in June 1998. On June 26 and 27, 1998, 77.1 mm of rain fell in Elk Island, only 7.1 mm less than the total monthly normal precipitation. This event took place one week after seeding and may have potentially eroded some seeds with the soil. The following month precipitation was below normal and almost half of it fell in one rainfall event. The final month of the 1998 growing season had conditions with only one third the normal monthly precipitation and 70% of it on one day. Lower precipitation than the normals occurred in 1999 and 2000 and the majority of rainfall events occurred in one or two days in each growing season month.

The combination of intermittent heavy rainfall followed by higher than normal temperatures may have been damaging to revegetation efforts as shown in other research. Bakker et al. (2003) state that the greatest challenge to prairie restoration is variability of precipitation. Erichsen-Arychuk et al. (2002) realized that drought conditions in disturbed areas would lead to more intense grazing. Singer et al. (1989) observed that decreased precipitation coupled with increased temperature during the growing season reduced grassland production by 50% in Yellowstone National Park. Georgiadis et al.

(1989) and Singer et al. (1989) observed that plants in the wild commonly grow well below maximum rates due to limitations of soil moisture. Martin and Chambers (2001) found that revegetating meadows with desirable native species required favorable weather conditions for seed germination and establishment. They observed that early seral and weedy species had higher establishment rates than seeded native species during dry, hot spring months.

2.5.2 Soil

Site 1 soils were loam to sandy loam textured with depth and Site 2 soils were loam to clay loam to clay textured with depth (Table 3). Site 1 soils were moderately alkaline and nonsaline (Table 4) with organic carbon contents ranging from 1.8 to 2 %. The total carbon was multiplied by 1.72 to determine the soil organic matter (Munshower 1994, Mapfumo 2003). Site 2 soils were neutral pH and nonsaline. Organic carbon ranged from 0.89 to 2.59%. No parameters were so high or low to cause major issues in revegetation. The low organic matter may have reduced water holding capacity and increased erosion potential, which coupled with the meteorological events, may have influenced plant establishment.

The topsoil was disturbed at both sites and most of the organic layer was removed during reclamation of the dumpsites. In some places the horizons were disturbed to a depth of 1 to 3 m (Kratko 1998) due to the differing size of refuse extracted at each site. Back filling and recontouring of the sites to predisturbed conditions further disrupted soil horizons. Soil from A, B and C horizons were incorporated together and mixed horizons were present at the surface during seedbed preparation. This may have inhibited plant establishment and growth as was also observed by Naeth et al. (1987) on pipeline rights-of-ways in Solonetzic native rangeland in southern Alberta.

Cleaning out the landfills mixed soil horizons at each site and as a result there were soil textural changes. Both sites had sand loss and there was an increase in silt and clay. Soil physical properties often present the main limitations to reclamation of disturbed lands (Naeth et al. 1991a). Soil texture indirectly influences plant root systems and plant growth through soil aeration, modification of soil water, erodibility and nutrient content and as root growth flourishes, it in turn affects soil texture (Naeth et al. 1991b,

Munshower 1994). Root growth is generally lower in coarse textured than fine textured soils due to lower fertility and lower water storage (Glinski and Lipiec 1990). The admixing at the sites was likely not sufficient to cause considerable revegetation concerns. However, the loss of topsoil may have impacted plant community development.

Munshower (1994) and Naeth (1999) indicated that a high pH, 8.0 and above, may lead to nutrient deficiencies for some plants. Soils with pH below 5.5 may contain high concentrations of metals and low levels of phosphorus that may inhibit plant growth. A study conducted by Naeth et al. (1987) observed pH levels rise to near 8 after soil disturbances. Soils that have a pH below 6.6 will require plants that are tolerant of acidic soils while plants tolerant to alkaline soils are necessary for pH levels above 7.3. Plants best suited to neutral soils are tolerant of pH values between 6.6 and 7.3. An EC of 4 dS m^{-1} or lower indicates a nonsaline soil while ECs > 4 dS m^{-1} denote increased levels of salt in the soil. It is unlikely that the soil pH and EC were in a range which would cause much concern in revegetation of the sites.

When vegetation is removed from soils they become susceptible to erosion. Visual observations throughout the course of the project detected some surface erosion and the formation of rills varying from approximately 1 to 5 cm in depth at the lower end of the slopes in each site. Arid soils in the disturbed areas that were newly seeded may have eroded after the occurrence of a major precipitation event during the first growing season. The rainfall event that occurred in 1998 had the potential to erode both seeds and soil. The combination of late seeding, low soil moisture and a major rainfall event may have hindered the revegetation process.

Although soil compaction was not measured on the site, the use of heavy equipment likely increased bulk density and penetration resistance. Soil compaction may also have occurred from grazing as was noted in numerous studies (Orr 1960, Laycock and Conrad 1967, Naeth et al. 1987, Donkor et al. 2002a). The higher clay content may also have facilitated compaction through grazing as indicated by Van Haveren (1983) and Chanasyk and Naeth (1995). The latter found that fine textured soil held more water than coarse textured soil and was therefore more susceptible to compaction especially if grazed when wet. Grazing may further affect compaction by removal of vegetation

which exposes soil to increased raindrop impact, decreases organic matter, increases surface evaporation, increases surface crust, decreases infiltration, increases erosion from runoff and reduces soil moisture (Naeth et al. 1990, Naeth and Chanasyk 1993).

Grazing may also impact soil chemical properties. Dung and urine deposition from ungulates can alter soil salinity (EC) (Chaneton and Lavado 1996) and soil acidity (pH) (Johnston et al. 1971). Mapfumo et al. (2000) observed significant increases in EC and pH on heavily grazed sites compared to medium or light grazing treatments.

Fencing disturbed areas to prevent herbivore access in the spring and summer may prevent the degradation of these disturbed areas, particularly via soil compaction. Fenced areas can be opened to wildlife access in the fall and winter when soil compaction is less likely. This has been shown by Naeth et al. (1990) where areas grazed at the end of the growing season were less compacted due to litter build up. Orr (1960) also noted that maximum compaction occurred in early to late summer.

2.5.3 Vegetation Response

2.5.3.1 Ground Cover

Although moss, trees, shrubs, rocks and dung were observed in the samples, the total combined percentages of all these variables totalled less than 5% (data not shown). Therefore only bare ground, litter and live vegetation were analyzed.

Bare ground decreased steadily from the first year of revegetation to the end of the study (Figures 1; Tables A13 through A16). At the end of the first year, bare ground was significantly lower in fenced than unfenced treatments. The highest bare ground was in unfenced broadcast seeded plots at 74.8% and lowest at 54.9% in fenced broadcast seeded plots. By the end of the second growing season all treatment combinations had increased bare ground except the unfenced broadcast seeded treatment, which stayed about the same. The greatest increase was 19.3% for the fenced drill seeded treatment. Overall the greatest increase was in the fenced treatments. This increase in the second growing season was unexpected but likely associated with the higher temperatures and lower precipitation in the previous growing season. By the end of the third growing season, bare ground had decreased considerably in all treatments, significantly so in the

fenced treatments. Seeding method had no significant effect on bare ground in all three growing seasons.

Litter steadily increased over the three years at both sites (Figure 1; Tables A13 through A16). At the end of the first growing season litter was similar in fenced versus unfenced treatments. By the end of the second growing season, unfenced treatments had more litter than fenced ones (not statistically significant), largely a result of increased weed litter. The litter in 1999 was higher in the unfenced plots because Site 2 unfenced plots had a large amount of Canada thistle which was going to seed at the time of sampling and was thus counted as litter. The fenced drill seeded treatment had the least litter at 5.7% while the unfenced broadcast seeded treatment had the most at 11.6%. In the final growing season the fenced plots had almost twice as much litter as the unfenced plots (statistically significant), as expected due to reduced animal utilization. The highest amount of litter was in the fenced broadcast seeded treatments at 36.7% and the lowest amount was 12% in the unfenced broadcast seeded treatment. Seeding method had no significant effect on litter cover in all three growing seasons.

Live vegetation increased from the first to the last growing season (Figure 1; Tables A13 through A16). Live vegetation in the fenced treatment was almost double (statistically significant) that in the unfenced treatment after the first growing season. The fenced broadcast seeded treatments had the highest live vegetation at 37.8% while the lowest was in the unfenced broadcast seeded treatment at 18.8%. By the end of the second season all treatment combinations experienced a drop in live vegetation with the lowest mean of 14% in the unfenced broadcast seeded treatment and the highest mean of 20.4% in the fenced broadcast seeded treatment. By the end of the final growing season live vegetation was higher than in any other year. Once again the fenced treatment had higher live vegetation than the unfenced treatment, but was not statistically significant. The fenced broadcast seeded mean was 50.7%, the highest value, and the lowest mean of 38.6% was in the unfenced drill seeded treatment. Seeding method had a significant effect only in the last year, with broadcast treatments having higher live vegetation than drilled.

These results are supported by Naeth et al. (1991a), who also found that grazed ranges had a decreased amount of live vegetation and litter cover and an increased

amount of bare ground. Bai et al. (2001) found bare ground was greater in grazed areas, while Gill et al. (1998) concluded that as grazing intensities increased bare ground increased, standing and fallen litter decreased and live vegetation cover decreased. Singer (1995) observed decreased litter in unfenced sites. Mapfumo et al. (2002) observed reduced amounts of litter in heavily grazed treatments compared to lightly grazed treatments and attributed it to the larger quantity of green herbage removed via grazing.

Contrary to what was found in this study, DePuit et al. (1980) found that broadcast seeding at twice the rate of drill seeding produced significantly greater grass canopy cover by the end of the second growing season. Doerr and Redente (1983) found that grass canopy cover in the second growing season on drill seeded plots was significantly greater than broadcast seeded plots seeded at twice the rate; however, these sites were fertilized and irrigated in the two growing seasons following seeding.

Fencing would significantly affect overall revegetation success as defined by decreasing bare ground and increased live vegetation and litter cover. Decreasing bare ground is of prime importance in the reduction of erosion and the establishment of a self-sustaining ground cover after revegetation. Thus, at least in the initial stages (up to three years) of revegetation, growing season grazing should be prevented.

2.5.3.2 Seeded Versus Non-Seeded Plant Density

Overall, non-seeded species had a greater density than seeded species in the final two growing seasons (Figures 2, A4, A5; Tables A15, A16). Both seeded and non-seeded species increased in all treatments up to the final growing season. The greatest increase in plant density from 1999 to 2000 occurred in the fenced broadcast seeded treatment. The smallest increase in plant density between 1999 and 2000 occurred in the fenced drill seeded treatment. In 1999, fencing and seeding method significantly affected seeded species density with seeded species increasing under fencing and broadcast seeding (Table A15). Plant density for non-seeded species was not significantly affected by either treatment in 1999. In 2000 plant density of seeded and non-seeded species was not significantly affected by fencing or seeding method treatments (Table A16).

The higher densities of non-seeded species in the early stages of plant community development were also reported by numerous researchers. Naeth (2000) observed that the dominance of weeds in the early stages of plant community development may be removed by the second growing season.

Disturbances can facilitate invasion of non-seeded species which can place adverse effects on the ecosystem by disrupting their processes and reducing their capacity to recover (Masters and Shelley 2001). Vitousek et al. (1996) proposed that enormous and continuous seed sources from human land-use practices facilitate invasions of non-native species. In a study in Elk Island National Park and other areas in central Alberta Vujnovic et al. (2002) observed that species richness and diversity of non-native plants displayed a significant increased relationship with the magnitude of the disturbance. They concluded increased severity of the disturbance on soil movement and compaction facilitated invasion of non-native species. Belcher and Wilson (1989) found that native plants can displace forage species, modify habitat structure and change plant-animal interactions within the ecosystem.

Desirable species can also invade and will enhance plant community diversity. Stark and Redente (1985) concluded that diversity helps the community to react to various environmental conditions. For example, if drought-like conditions persisted, as they did in this study, having plants in the seed mixture that can cope with drier conditions will reduce erosion. Plant species diversity will either increase or decrease in response to grazing, depending on initial conditions, intensity and duration of the grazing and how selective the grazers are towards the plant species (Harper 1969, Vinton et al. 1993). Vujnovic et al. (2002) stated that conservation area managers may want to allow some light grazing from wildlife to increase native species diversity and observational evidence suggests that a 20 year hiatus from grazing may be required for native species to reach dominance in disturbed areas.

Revegetating disturbed areas may prevent non-native species from dominating the plant community and prevent the landscape from being occupied by a small, aggressive fraction of the world's plant species (Hobbs and Humphries 1995, Simberloff 2001, Olliff et al. 2001). Naeth (1997) concluded that even though weedy species are undesirable

because they provide poor forage and compete with grass seedlings for nutrients and water, they reduce bare ground and trap snow after disturbances, thus reducing erosion.

2.5.3.3 Plant Biovolume

Species biovolume of seeded and non-seeded species among treatments did not consistently increase or decrease from 1999 to 2000 and numerical changes were small in all cases (Figures 3, A6, A7; Tables A15, A16). Overall, biovolume of seeded species was slightly higher in fenced than unfenced treatments in 1999 and 2000. Non-seeded species were slightly greater in unfenced than fenced treatments in both years. Plant biovolumes of seeded and non-seeded species were not significantly different with any of the fencing or seeding treatments in any given year.

Biovolume may have been greater in the unfenced drill seeded treatments in 1999 due to the high traffic of ungulates. The entire unfenced treatments were virtually trampled and this may have pushed the drilled seeds deeper than the initial seeding depth of 2.54 cm. The increased depths may have prevented emergence.

2.5.3.4 Plant Biomass

In 2000, plant biomass statistical analyses were conducted between fenced and unfenced treatments as well as between range cages and unfenced treatments. No analyses were conducted between range cages and fenced treatments because the range cages were applied to areas that were grazed in 1998 and 1999 and the fenced treatments were not grazed in any year.

Total plant biomass was greatest in the fenced treatment, followed by the range cages and the unfenced treatment (Figure 4; Table A16). Biomass was not significantly affected by seeding method. However, there was statistical significance with fencing and range cages. The statistical analysis completed on the biomass of the range cages and the unfenced treatment detected no significance with seeding method treatment. There was a significant difference between unfenced and caged treatments indicating grazing had occurred on the sites.

Biomass was highest in the fenced areas because fencing prevented wildlife from affecting the vegetation. The cages were second in total biomass since they prevented wildlife from grazing the vegetation to the ground level, yet allowed wildlife to graze the

vegetation that projected through the cages. Since the unprotected areas were submitted to grazing, the probability that randomly selected areas would be grazed to ground level greatly increased.

Other researchers found similar results to this study. Bai (2001) found heavy grazing reduced native grass biomass while moderate grazing reduced or had no effect on biomass and light grazing had no effect. Defoliation did not decrease above ground biomass in wet years, but in dry years biomass was severely reduced due to decreased tiller density (Fahnestock and Detling 1999). Bork et al. (2001) concluded upland herbage production in Elk Island National Park is directly dependent on precipitation, particularly that which falls during the growing season (May to August). Jackson (1999) observed significantly higher biomass in drill seeded than broadcast seeded plots in the second and third growing seasons. Werner and Urness (1998) concluded that not only elk herbivory but the regrowth ability of vegetation in the areas and environmental conditions associated with plant defoliation responses also affected biomass.

2.5.3.5 Plant Species Composition

Most of the seeded species remained in the plant community throughout the last two growing seasons and none were affected by seeding method or fencing treatment (Table 5, A15, A16). The wheatgrasses were much more successful at establishing than the other species, with nodding brome and green needle grass being the least successful. Nodding brome may not have been successful due to a low percentage of pure live seed. An error occurred in the green needle grass calculation and as a result, each subplot was short approximately 1.1 g of seed. Individual seeded species responded differently between 1999 and 2000. Slender and streambank wheatgrass increased considerably in all treatments, while western wheatgrass followed a similar pattern but remained constant in the fenced broadcast treatment. Slender wheatgrass dominated all treatments except the unfenced drill seeded treatment in 2000. Wheatgrass domination in seeded plant communities in reclaimed sites is quite common (Naeth 2002 personal communication).

Most non-seeded species had less than 10 plants m^{-2} (Table 6). Those species with more than 10 plants m^{-2} included alsike clover, Canada thistle, dandelion, horsetail, Kentucky bluegrass, perennial sow thistle, plantain, white clover and unidentified weed

seedlings. All are considered weeds or relatively aggressive native species and all were predominant in all treatments. Many species were found in only one of the treatments. Some of the desired species that moved into the sites, in many of the treatments, included American vetch, Canada goldenrod, cinquefoil, curled dock and yarrow. These are common species in revegetated sites. Common weed species included Canada thistle, dandelion, narrow leaved hawksbeard and perennial sow thistle as well as numerous weed seedlings that were not identified due to their very small size. Canada thistle and perennial sow thistle are noxious weeds that may require further management action such as the application of herbicide.

Desired non-seeded species were considerably lower in the unfenced broadcast treatment than in any of the other treatments in both years (Table 6). This was also the treatment that was dominated by slender wheatgrass and had high amounts of weeds like dandelion and invader species like white clover.

Total number of plants was higher in 2000 than 1999 in all treatments (Tables 5 and 6). Species richness of non-seeded plants was similar in all treatments ranging from 15 species in the unfenced broadcast seeded treatment (2000) to 25 species in the 1999 and 2000 unfenced drill seeded treatments (Table 6). Species richness was the same for seeded species in both years and all treatments. It was highest in the unfenced drilled treatment and lowest in the unfenced broadcast treatment in both 1999 and 2000 for non-seeded species. None of these differences were statistically significant.

Oba et al. (2001) similarly found that species richness was not significantly greater in ungrazed than grazed areas and long-term exclusion from grazing did not increase species richness in arid areas. Stohlgren et al. (1999b) found no significant difference in native species richness between fenced and unfenced areas and combined native and non-native species richness was not significantly different between fencing treatments. Jackson (1999) observed greater numbers of native species in drill seeded treatments; however, they were not significantly different at the end of the first growing season.

Contrary to this study, Depuit and Coenenberg (1979) found that broadcast seeding native plant species at twice the rate of drill seeding on topsoiled mine spoils was superior in terms of productivity, stand composition and diversity of seeded species.

However, that project used fertilization as a treatment and drill seeding was superior to broadcast seeding when seeding rates were equal. At equal seeding rates large seeded species dominated in drill seeded treatments while broadcast seeding was dominated by small seeded species.

Although this study can be used to assess species invasion with the small diversity mix used, it can not be used to specifically indicate how individual species in the mix affected invasion rates of specific non-seeded species. Stohlgren et al. (1999a) stated that trying to understand causes and patterns of native species richness in an attempt to explain variation in plant communities would require consistent observations of invasibility patterns that cross several spatial scales and the inclusion of non-native species may further complicate the situation.

2.5.4 WILDLIFE

Numerous signs of wildlife were found around the perimeter of the fenced areas and in the adjacent unfenced areas (Table 7). An observation in February 2000 detected that a portion of the fence had been pushed in approximately 1 m between two of the fences supporting posts. No other wildlife impacts beyond the indentation of the fence were observed and these impacts did not affect the fenced plots.

In the final two growing seasons, the combined total area of all wildlife interactions at both sites was 1558 m^2. The combined total area affected by wildlife in 1999 was 974 m^2 and decreased to 584 m^2 in 2000. The total area of the plots affected in the unfenced broadcast seeded and unfenced drill seeded treatments from both sites in 1999 and 2000 was 785 and 773 m^2, respectively. Four of the indicators decreased and one of them increased from 1999 to 2000.

2.5.4.1 Bedding and Trampling

Bedding by ungulates was the only indicator to increase from the second growing season to the third, going from 3 m^2 in 1999 to 31 m^2 in 2000. Over the two years, the area trampled by ungulates was 978 m^2. Every square meter of both seeding treatments in the unfenced plots at Site 1 in 1999 had evidence of tracks from ungulates. Site 2 was

similar to Site 1 with all but 1 m^2 of the unfenced plots being affected in 1999 and decreasing by 2000.

As time progressed, ground cover increased and bare ground decreased. Due to this trend, evidence of wildlife interactions, such as tracks, became undetectable. Arid conditions were another factor that may have decreased detection of trampling. When the surface lost moisture and the ungulates utilized the unfenced plots, the hooves did not make deep impressions. Wind erosion and/or a precipitation event may also have erased evidence of trampling.

Trampling of the seeded areas created depressions, which had the potential to push seeds well below desired seeding depths and inhibit seedling emergence. When soil moisture was low, depressions were no greater than 2 to 3 cm; however, they exceeded 10 cm when soil moisture was high. Compaction intensified in the first growing season after the two days of heavy rainfall. Physical soil properties are altered from animal hooves, which increase soil bulk density (Wood and Blackburn 1981). Warren et al. (1986) found that hoof action reduced the size of naturally occurring soil aggregates and increased bulk density at the surface horizon. Up to 200 kPa of pressure is potentially exerted from animal hooves and this value is greater than a tractor, which can range from 30 to 150 kPa (Profitt et al. 1993). The former study dealt with sheep, which weigh considerably less than the larger ungulates found in Elk Island National Park. Naeth et al. (1990) reported increased bulk densities to a depth of 7.5 cm due to trampling under heavy grazing intensities and Donkor et al. (2002b) reported compaction to a depth of 10 cm. The effects of animal trampling may be reduced through natural processes such as freeze-thaw action, wet-dry cycles and earthworm activity (Mapfumo et al. 1999).

2.5.4.2 Grazing

Grazing was the second highest observed interaction affecting a combined total of 418 m^2 between the two sites in 1999 and 2000. Grazing decreased from 1999 to 2000 at both sites and the greatest amount of grazing occurred in the broadcast seeded plots.

In 2000, the introduction of range cages to assess biomass inhibited wildlife from interacting with the ground covered by the cages. There was 27 m^2 of ground covered at each site. The sizable decrease in total affected area from 1999 to 2000 (974 to 584 m^2)

can partly be attributed to the introduction of range cages in 2000 and the no evidence parameter at both sites.

The literature contains numerous reports regarding the effects cattle and other domesticated livestock place on soil physical and chemical properties, plant physiology and plant communities, however, there is limited research on the effects native ungulates have on these parameters. There is even less information on the above stated parameters in a revegetated previously disturbed range.

Spatial and temporal patterns of grazing by native ungulates regulate grassland function (Frank and McNaughton 1993, Hobbs 1996, Augustine and Frank 2001, Augustine and McNaughton 1998, Coughenour 1991). Ungulates often select the landscape that contains the greatest amount of resources, then the most productive community contained in the landscape and finally the most palatable species in a feeding station, an area an ungulate can feed without moving its feet (Adler et al. 2001). Grazing in the early part of the growing season when soil moisture is high and nutrients are readily available give vegetation the resources and time to regrow and recover. With grazing in the early season, from March to April, plant regrowth was rapid; however, with later grazing, from May to June, plant regrowth was much slower (Richards and Caldwell 1985). Grazing of standing dead biomass in winter when plants were dormant stimulated productivity in the following growing season (Frank and McNaughton 1993).

McNaughton (1979) put forth the "grazing optimization hypothesis" which states grass production increases as grazing increases from zero to moderate intensities, but if grazing intensity becomes excessive, grass production decreases to a level below ungrazed grasses. Preventing continuous grazing over the entire growing season may help reduce or eliminate the damaging effects of grazing on soil properties (Chanasyk and Naeth 1995). Donkor et al. (2002a), studying sites with Dark Gray Luvisolic soil in the Boreal Mixedwood forest, similar to soils in Elk Island National Park, observed that ungrazed range had more green herbage, standing dead and fallen litter than range which had been grazed either continuously or in a short duration system. Grazers such as elk (*Cervus elaphus*) and bison (*Bos bison bison*) can stimulate grassland aboveground production. Plant production responses to defoliation can vary widely, contingent upon a complex interaction of ecological factors (Georgiadis et al. 1989).

Large native ungulates have indirect effects on nutrient and energy flows in grassland habitats that go well beyond the direct impacts of the removal of plant tissue. Adler et al. (2001) researched the effects of grazing on spatial heterogeneity of vegetation, which in turn influenced ecosystem processes and biodiversity. Grazing also lowers the plants ability to transpire which increases soil moisture and water use efficiency. The amount of available soil moisture prior, during and after grazing events determines aboveground plant production. Grazing also lowers the amount of standing dead biomass and litter, thus enabling greater light penetration, which in turn increases soil temperature, which may assist in decomposition. Defoliation of plants opens the canopy to light thus increasing the potential for photosynthesis. The higher photosynthetic rate of regrowth can lead to higher growth rates for grazed plants compared to ungrazed plants.

After grazing by herbivores, greater photosynthate goes to aboveground growth, which in turn reduces root growth. Grazing can reduce root activity, which can in turn affect soil bulk density (Naeth et al. 1991a). As grazing intensity increases, more shallow-rooted species replace deep-rooted species (Naeth et al. 1991a). Other studies have shown grazing does not affect root growth (Frank and McNaughton 1993). However, in cases where root growth has shown to be affected there is a reduction in the carbon:nitrogen ratio of organic material and an increase in net mineralization.

Plants displaying symptoms of stress are likely to increase production following grazing (Hilbert et al. 1981). Total above ground grass production (the sum of foliage removed by grazing ungulates plus the compensatory growth of plants) is maximized at moderate grazing intensities (Georgiadis et al. 1989). A study conducted by Richards and Caldwell (1985) observed severe clipping rates of *Agropyron* spp. reduced carbohydrate reserves by more than 40% and that apical meristem removal from grazing sharply reduced regrowth efficiency due to a reduced rate of regrowth production. A minimum quantity of forage is necessary to ensure that ungulates will regraze a previously grazed patch (Adler et al. 2001), different amounts in the controls may prevent wildlife from returning to as was the case from 1999 to 2000. Grazing reduces the amount of available forage, but in many ecosystems, it increases nitrogen or crude protein (Jeffries et al. 1994, Holland and Detling 1990). When grazing changes species

composition to unpalatable or low nitrogen plant species, patch grazing theoretically should not persist (Jeffries et al. 1994).

Grazing may have decreased due to a greater amount of non-seeded unpalatable species dominating the sites. Broadcast seeded plots may have had more grazing because of the higher seeding rate and the greater number of palatable seeded species.

2.5.4.3 Scat

Wildlife scat decreased from the second to third growing season. In 1999 the combined amount of scat from both sites was observed at 69 m^2 and by 2000, this dropped to only 31 m^2. A decrease in scat is directly linked to a decrease in usage from ungulates.

Herbivores graze nutrient poor material and excrete nutrient rich material that is readily available for plants and soil microorganisms (Frank and McNaughton 1993). Scat from ungulates also had the potential to increase the introduction of non-seeded species into the disturbed sites. Dung contains seeds from previously grazed plant species and can also provide a source of nutrients which benefits plant emergence and establishment. However, revegetation projects may be negatively impacted if dung contains seeds from non-native or invasive plant species, which have the potential to grow faster than native plants. Seeds that are weedy or invasive can encumber revegetative projects.

2.5.4.4 Wallows

The area affected by bison wallows decreased in the final growing season. There were two wallows totalling 28 m^2 at Site 2. The largest one covered approximately 24 m^2 and the smaller one was approximately 4 m^2 at the southern most tip of the site. In 1999, bison used the wallow heavily as was evident by the lack of vegetation inside of it. In 2000, weedy species dominated the inside of the wallow. This influx of weedy species was another indication of a decreased presence of ungulates in the last assessment year. If bison would have been utilizing the wallow in 2000, vegetation would not have been present because vegetation is removed when bison wallow. The creation of the wallow displaced or entirely removed the seeds from revegetation and prevented the emergence

of the seeded species in the final growing season. The wallow also facilitated the invasion of non-seeded species.

A small amount of research on wallows has been undertaken. Coppedge et al. (1999) observed that bison (*Bos bison bison*) created wallows in sandy loam soils. Wallowing by bison may also result in unique patterns of environmental patchiness and plant responses (Damhoureyeh and Hartnett 1997). Both of these studies found that vegetation would not persist in the wallows when they were actively being utilized by bison.

2.5.4.5 No Evidence of Wildlife Use

In this study, "no evidence" was defined as none of the above five indicators being present. Range cages prevented wildlife from trampling the ground therein. Total area with no evidence of use by ungulates was the third highest observed parameter in 2000. Wildlife were present in the area because there were indicators in adjacent 1 m^2; however, ground cover prevented wildlife from imprinting the ground. In 1999, every square meter was affected by ungulates; however, by 2000 160 m^2 had no evidence of ungulate utilization. Only 6 m^2 showed no evidence at Site 1 with the remainder of no evidence occurring at Site 2.

The wildlife indicator of no evidence was not a factor in 1999 because there was an excessive amount of bare ground at both sites. The lack of vegetation and litter in 1999 at both sites facilitated observing tracks on the bare ground. As time progressed, new plant growth and deposition of litter from weedy species covered the ground and it prevented wildlife from leaving tracks. The combined effects of decreased grazing and trampling established the need for a no evidence parameter.

2.5.5 Suitability of Plants Selected in the Revegetation Seed Mix

From the perspective of selecting plants suitable for revegetation, it was crucial to meet certain criteria. Adherence to National Park guidelines state that all revegetation efforts must use native plant species, which was the single most important criteria warranting strict attention. Ecological setting, adaptation to disturbance, competitive ability, moisture preference, drought tolerance, winter hardiness, soil preference, pH

tolerance, salt tolerance, erosion control, reaction to wildlife interactions, palatability and seed availability were other factors influencing seed mix selection. Plants had to be grazing tolerant species to meet the end goal of the revegetation project which was the creation of usable habit for ungulates.

In this study the seeded plant species, other than nodding brome and green needle grass, performed acceptably. Plant species selected were all native species found in the transition zone between the Aspen Parkland-Lower Boreal Mixedwood ecoregions (Moss 1983). This revegetation project required plants suited to loam soils that were neutral to moderately alkaline in pH. The species selected did not have to be salt tolerant because EC was below 1.0 dS m^{-1}.

The time of seeding may have prevented vegetation from establishing due to the shorter growing season. The sites were seeded in mid-June and cool season species usually are established most easily by seeding in the spring or late fall (Vogel 1987).

2.6 MANAGEMENT IMPLICATIONS

Fencing revegetated areas in EINP significantly decreased bare ground as well as significantly increased litter. Fencing increased plant biomass and plant biovolume of seeded species. Unfenced areas will facilitate a larger percentage of non-seeded species compared to fenced treatments. Park mangers should fence newly revegetated areas and prevent wildlife access during the growing season. Fenced areas can be opened to wildlife access in the fall and winter when soil compaction is less likely.

By the end of the third growing season, broadcast seeding produced significantly more vegetation than drill seeding. The density and biovolume of seeded species is not significantly different for either seeding treatment. Park mangers will have to consider accessibility of disturbed areas as well as the economical feasibility of both seeding treatments.

Overall the majority of seeded species in the seed mix were present in the revegetated plots. Nodding brome and green needle grass may need to be reconsidered as suitable species to put in the seed mixture due to their performance. Biovolume of non-seeded species were greater than seeded species in all treatment combinations. Park

mangers may want to consider spraying herbicides that are selective to weedy species as well as being low in mammalian toxicity to prevent the dominance of undesirable species.

2.7 CONCLUSIONS

- Overall broadcast versus drill seeding had no significant effect on the vegetation parameters assessed, however, by the end of the study live vegetation was significantly greater in broadcast seeded plots.
- Overall fencing had a significant impact on the vegetation parameters assessed. This was particularly noteworthy with bare ground, which decreased significantly with fencing in two out of the three study years.
- Wildlife heavily utilized the research area and fencing prevented the impact of that utilization.
- The seeded wheatgrass species and fringed brome showed potential for revegetation; green needle grass and nodding brome either did not survive or did so in very small numbers and did not increase with time.
- The seed mix did not prohibit invasion by non-seeded species. Many non-seeded species grew in the sites; some of them were weeds and some of them were desired native species.

2.8 REFERENCES

Adler, P.B., D.A. Raff and W.K. Lauenroth. 2001. The effect of grazing on the spatial heterogeneity of vegetation. Oecologia 128:465-479.

AGRASID. 1998. Agricultural region of Alberta soil inventory database. Version 1.0. Alberta Agriculture, Food and Rural Development. Edmonton AB.

Agriculture Canada Expert Committee on Soil Survey. 1987. The Canadian system of soil classification. 2nd ed. Agric. Can. Publ. 1646. 164 pp.

Archer, S. and D.A. Pyke. 1991. Plant-animal interactions affecting plant establishment and persistence on revegetated rangeland. J. Range Manage. 44:558-565.

Augustine, D.J. and D.A. Frank. 2001. Effects of migratory grazers on spatial heterogeneity of soil nitrogen properties in a grassland ecosystem. Ecology (82):3149-3162.

Augustine, D.J. and S.J. McNaughton. 1998. Ungulate effects on the functional species

composition of plant communities: herbivore selectivity and plant tolerance. J. Wildl. Manage. 62:1165-1183.

Bai, Y., Z. Abouguendia and R.E. Redmann. 2001. Relationship between plant species diversity and grassland condition. J. Range Manage. 54:177-183.

Bakker, J.D., S.D. Wilson, J.M. Christian, X.Li, L.G. Ambrose and J. Waddington. 2003. Contingency of grassland restoration on year, site and competition from introduced grasses. Ecol. Appl. 13:137-153.

Belcher, J.W. and S.D. Wilson. 1989. Leafy spurge and the species composition of a mixed grass prairie. J. Range Manage. 42:172-175.

Blenis, P.V. 2000. Post-spacing mortality of lodgepole pine from armillaria root disease. For. Chron. 76(5):753-757.

Blenis, P.V. 1999. Personal communication. Statistical advisor, Department of Renewable Resources, University of Alberta. Edmonton AB.

Bork, E.W., T. Tamiko and B. McDougall. 2001. Herbage response to precipitation in central Alberta boreal grasslands. J. Range Manage. 54:243-248.

Bork, E.W., R.J. Hudson and A.W. Bailey. 1997. *Populus* forest characterization in Elk Island National Park relative to herbivory, prescribed fire and topography. Can. J. Bot. 75:1518-1526.

Bostock, H.S. 1970. Physiographic subdivisions of Canada. In: Geology and Economic Minerals of Canada. R.J.W. Douglas (ed.) Geologic Survey of Canada. Economic Geology Report No. 1. Pp. 9-31.

Boyce, M.S. 1998. Ecological-process management and ungulates: Yellowstone's conservation paradigm. Wildl. Soc. Bull. 26:391-398.

Chanasyk, D.S. and M.A. Naeth. 1995. Grazing impacts on bulk density and soil strength in the foothills fescue grasslands of Alberta, Canada. Can. J. Soil Sci. 75:551-557.

Chaneton, E.J. and R.S. Lavado. 1996. Soil nutrients and salinity after long-term grazing exclusion in a Flooding Pampa grassland. J. Range Manage. 49:182-187.

Chapman, R.J. 2003. Personal communication. Park Warden/Conservation Biologist, Elk Island National Park. Fort Saskatchewan AB.

Chapman, R.J. 1991. The discoverer's guide to Elk Island National Park. Lone Pine Publishing. Edmonton AB. 96 pp.

Cool, N. 2003. Personal communication. Park Warden/Wildlife Biologist, Elk Island National Park. Fort Saskatchewan AB.

Coppedge, B.R., S.D. Fuhlendorf, D.M. Engle and B.J. Carter. 1999. Grassland soil depressions: relict bison wallows or inherent landscape Heterogeneity? Am. Mid. Nat. 142:382-392.

Coughenour, M.B. 1991. Spatial components of plant-herbivore interactions in pastoral, ranching and native ungulate ecosystems. J Range Manage. 44:530-542.

Crown, P.H. 1977. Soil survey of Elk Island National Park. Alberta Institute of Pedology. Edmonton AB. 128 pp

Damhoureyeh, S.A. and D.C. Hartnett. 1997. Effects of bison and cattle on growth, reproduction and abundances of five tallgrass prairie forbs. Am. J. Bot. 84:1719-1728.

DePuit, E.J., J.G. Coenenberg and C.L. Skilbred. 1980. Establishment of diverse native plant communities on coal surface mined lands in Montana as influenced by

seeding method, mixture and rate. Montana Agric. Exp. Stn. Res. Rep. 163. 64 pp.
DePuit, E.J. and J.G. Coenenberg. 1979. Methods for establishment of native plant communities on topsoiled coal stripmine spoils in the northern Great Plains. Reclam. Rev. 2:75-83.
Doerr, T.B. and E.F. Redente. 1983. Seeded plant community changes on intensively disturbed soils as affected by cultural practices. Reclam. Reveg. Res. 2:13-24.
Donkor, N.T., J.V. Gedir, R.J. Hudson, E.W. Bork, D.S. Chanasyk and M.A. Naeth. 2002a. Impacts of grazing systems on soil compaction and pasture production in Alberta. Can. J. Soil Sci. 82:1-8
Donkor, N.T., E.W. Bork and R.J. Hudson. 2002b. *Bromus-Poa* response to defoliation intensity and frequency under three soil moisture levels. Can. J. Plant Sci. 82:365-370.
Douglas, R.J.W. 1970. Geology and economic minerals of Canada. Energy, Mines and Resources Canada, Econ. Geol. Report 1. Ottawa ON. 838 pp.
Environment Canada. 2003a. Canadian monthly climate data and 1971-2000 normals. CD-rom. Ottawa ON.
Environment Canada. 2003b. 1998-2000 monthly climate data. Edmonton AB.
Erichsen-Arychuk, C., E.W. Bork and A.W. Bailey. 2002. Northern dry mixed prairie responses to summer wildfire and drought. J. Range Manage. 55:164-170.
Fahnestock, J.T. and J.K. Detling. 1999. Plant responses to defoliation and resource supplementation in the Pryor Mountains. J. Range Manage. 52:263-270.
Frank, D.A. and S.J. McNaughton. 1993. Evidence for the promotion of aboveground grassland production by native large herbivores in Yellowstone National Park. Oecologia 96:157-161.
Gill, S.I., M.A. Naeth, D.S. Chanasyk and V.S. Baron. 1998. Runoff and sediment yield from snowmelt and rainfall as influenced by forage type and grazing intensity. Can. J. Soil Sci. 78:699-706.
Georgiadis, N.J., R.W. Ruess, S.J. McNaughton and D. Western. 1989. Ecological conditions that determine when grazing stimulates grass production. Oecologia 81:316-322.
Glinski, J. and J. Lipiec. 1990. Soil physical conditions and plant roots. CRC Press Inc. Boca Raton FL. 250 pp.
Harper, J.L. 1969. The role of predation in vegetation diversity. In: Brookhaven Symposium in Biology. Brookhaven National Laboratory. Upton NY. No.22:48-62.
Hilbert, D.W., D.M. Smith, J.K. Detling and M.I. Dyer. 1981. Relative growth rates and the grazing optimization hypothesis. Oecologia 51:14-18.
Hobbs, R.J. and S.E. Humphries. 1995. An integrated approach to the ecology and management of plant invasions. Conserv. Biol. 9:761-770.
Hobbs, N.T. 1996. Modification of ecosystems by ungulates. J. Wildl. Manage. 60:695-712.
Holland, E.A. and J.K. Detling. 1990. Plant response to herbivory and belowground nitrogen cycling. Ecology 71:1040-1049.
Jackson, L.L. 1999. Establishing tallgrass prairie on grazed permanent pasture in the upper Midwest. Restor. Ecol. 7:127-138.
Jeffries, R.L., D.R. Klein and G.R. Shaver. 1994. Vertebrate herbivores and northern

plant communities: reciprocal influences and responses. Oikos 71:193-206.
Jennings, D.B. 1983. The Quaternary geomorphology of Elk Island National Park, central Alberta. M.Sc. thesis. University of Alberta. Edmonton AB.
Johnston, A., J.F. Dormaar and S. Smoliak. 1971. Long-term grazing effects on fescue grassland soils. J. Range Manage. 24:185-188.
Kerr, D.S., L.J. Morrison and K.E. Wilkinson. 1993. Reclamation of native grasslands in Alberta: A review of the literature. Alberta Land Conservation and Reclamation Council Report No. 93-1. ISBN 0-7732-0881-X. Edmonton AB. 205 pp.
Kratko, H. 1998. Personal communication. Kratko Demolition and Excavating. Ardrossan AB.
Laycock, W.A. 1967. How heavy grazing and protection affect sagebrush-grass ranges. J. Range Manage. (20):206-213.
Laycock, W.A. and P.W. Conrad. 1967. Effect of grazing on soil compaction as measured by bulk density on a high elevation cattle range. J. Range Manage. 20:136-140.
Mapfumo, E. 2003. Personal communication. Department of Renewable Resources, University of Alberta. Edmonton AB.
Mapfumo, E., M.A. Naeth, V.S. Baron, A.C. Dick and D.S. Chanasyk. 2002. Grazing impacts on litter and roots: perennial versus annual grasses. J. Range Manage. 55:16-22 pp.
Mapfumo, E., D.S. Chanasyk, V.S. Baron and M.A. Naeth. 2000. Grazing impacts on selected soil parameters under short-term sequences. J. Range Manage. 53:466-470.
Mapfumo, E., D.S. Chanasyk, M.A. Naeth and V.S. Baron. 1999. Soil compaction under grazing of annual and perennial forages. Can. J. Soil Sci. 79:191-199.
Martin, D.W. and J.C. Chambers. 2001. Restoring degraded riparian meadows: biomass and species response. J. Range Manage. 54:284-291.
Masters, R.A. and R.L. Shelley. 2001. Invited synthesis paper: Principles and practices for managing rangeland invasive plants. J. Range Manage. 54:502-517.
McKeague, J.A. 1978. Manual on soil sampling and methods of analysis. 2nd ed. Can. Soc. Soil Sci., Ottawa ON. 212 pp.
McNaughton, S.J. 1979. Grazing as an optimization process: grass-ungulate relationships in the Serengeti. Am. Nat. 113:691-703.
Milliken, G.A. and D.E. Johnson. 1984. Analysis of messy data: Volume 1: Designed experiments. Van Nostrand Reinhold Company Inc. New York N.Y. 473 pp.
Montalvo, A.M., P.A. McMillan and E.B. Allen. 2002. The relative importance of seeding method, soil ripping and soil variables on seeding success. Restor. Ecol. 10:52-67.
Moss, E.H. 1983. Flora of Alberta. University of Toronto Press. Toronto ON. 687 pp.
Munshower, F.F. 1994. Practical handbook of disturbed land revegetation. Lewis Publishers, Boca Raton FL. 265 pp.
Naeth, M.A. 2002. Personal communication. Department of Renewable Resources, University of Alberta. Edmonton AB.
Naeth, M.A. 2000. Native species establishment for resource conservation. The Alberta Agricultural Research Institute. 42 pp.
Naeth, M.A. 1999. Renewable Resources 475-Vegetative reclamation, Revegetation and

Restoration. Class lecture notes. Dept. of Renewable Resources, University of Alberta. Edmonton AB.

Naeth, M.A. 1997. The long-term impact of pipeline construction on Solonetzic mixed prairie. NOVA Gas Transmission Ltd., Community Resources. Edmonton AB. 55 pp.

Naeth, M.A. and D.S. Chanasyk. 1995. Grazing effects on soil water in Alberta foothills fescue grasslands. J. Range Manage. 48:528-534.

Naeth, M.A., D.S. Chanasyk and M.W. Dell. 1995 . The impact of forages on soil physical properties. Alberta Agricultural Research Institute. Edmonton AB. 60 pp.

Naeth, M.A. and D.S. Chanasyk. 1993. Impacts of grazing on rangeland hydrology in the foothills fescue grasslands. Alberta Agriculture. Edmonton AB. 59 pp.

Naeth, M.A., A.W. Bailey, D.J. Pluth, D.S. Chanasyk and R.T. Hardin. 1991a. Grazing impacts on litter and soil organic matter in mixed prairie and fescue grassland ecosystems of Alberta. J. Range Manage. 44:7-12.

Naeth, M.A., D.J. White, D.S. Chanasyk, T.M. Macyk, C.B. Powter and D.J. Thacker. 1991b. Soil physical properties in reclamation. Alberta Land Conservation and Reclamation Council Report No. RRTAC 91-4. ISBN 0-7732-0880-1. 216 pp.

Naeth, M.A., A.W. Bailey, D.S. Chanasyk and D.J. Pluth. 1991c. Water holding capacity of litter and soil organic matter in mixed prairie and fescue grassland ecosystems of Alberta. J. Range Manage. 44:13-17.

Naeth, M.A., D.J. Pluth, D.S. Chanasyk, A.W. Bailey and A.W. Fedkenheuer. 1990. Soil compacting impacts of grazing in mixed prairie and fescue grassland ecosystems of Alberta. Can. J. Soil Sci. 70:157-167.

Naeth, M.A., D.S. Chanasyk, W.B. McGill, A.W. Bailey and R.T. Hardin. 1988. Changes in soil water regime after pipeline construction in Solonetzic mixed prairie rangeland. Can. J. Soil Sci. 68:603-610.

Naeth, M.A., W.B. McGill and A.W. Bailey. 1987. Persistence of changes in selected soil chemical and physical properties after pipeline installation in Solonetzic native rangeland. Can. J. Soil Sci. 67:747-763.

Oba, G., O.R. Vetaas and N.C. Stenseth. 2001. Relationships between biomass and plant species richness in arid-zone grazing lands. J. Appl. Ecol. 38:836-845.

Olliff, T., R. Renkin, C. McClure, P. Miller, D. Price, D. Reinhart and J. Whipple. 2001. Managing a complex exotic vegetation program in Yellowstone National Park. West. N. Am. Nat. 61:347-358.

Orr, H.K. 1960. Soil porosity and bulk density on grazed and protected Kentucky Bluegrass range in the Black Hills. J. Range Manage. 13:80-86.

Porter, W.F. and H.B. Underwood. 1999. Of elephants and blind men: deer management in the U.S. national parks. Ecol. Appl. 9:3-9.

Profitt, A.P.B., S. Bendotti, M.R. Howell and J. Eastham. 1993. The effect of sheep trampling on soil physical properties and pasture growth for a red-brown earth. Aust. J. Agric. Res. 44:317-331.

Richards, J.H. and M.M. Caldwell. 1985. Soluble carbohydrates, concurrent photosynthesis and efficiency in regrowth following defoliation: a field study with *Agropyron* species. J. Appl. Ecol. 22:907-920.

Sheldrick, B.H. and C. Wang. 1993. Particle size distribution. In: Soil sampling and methods of analysis. M.R. Carter, ed. Lewis Publishers, Boca Raton FL. Pp.

499-511.
Simberloff, D. 2001. Biological invasions-how are they affecting us, and what can we do about them? West. N. Am. Nat. 61:308-315.
Singer, F.J. 1995. Effects of grazing by ungulates on upland bunchgrass communities of northern winter range of Yellowstone National Park. Northwest Sci. 69:191-203.
Singer, F.J., W. Schreiner, J. Oppenheim and E.O. Garton. 1989. Drought, fires and large mammals-evaluating the 1988 severe drought and large scale fires. Bioscience 39:716-722.
Stark, J.M., and E.F. Redente. 1985. Soil-plant diversity relationships on disturbed site in northwestern Colorado. Soil Sci. Soc. Am. J. 49:1028-1034.
Steel, R.G.D., J.H. Torrie and D.A. Dickey. 1997. Principles and procedures of statistics: A biometrical approach. McGraw Hill. New York N.Y. 666 pp.
Stohlgren, T.J., D. Binkley, G.W. Chong, M.A. Kalkhan, L.D. Schell, K.A. Bull, Y. Otsuki, G. Newman, M. Bashkin and Y. Son. 1999a. Exotic plant species invade hot spots of native plant diversity. Ecol. Monogr. 69(1)25-46.
Stohlgren, T.J., L.D. Schell and B. Vanden Heuvel. 1999b. How grazing and soil quality affect native and exotic plant diversity in Rocky Mountain grasslands. Ecol. Appl. 9:45-64.
Strachan, J.E., C.B. Blyth, D. Graham, K. Schroeder, N. Cool, D. Gilbride, B. Higgins, R. McGill and S. McNaughton. 1986. Elk Island National Park, Resource description and analysis: Volume 1. Parks Canada, Western Region. 428 pp.
Strong, W.L. and K.R. Leggat. 1992. Ecoregions of Alberta. Alberta Forestry, Lands and Wildlife. Edmonton AB. 59 pp.
Van Haveren, B.P. 1983. Soil bulk density as influenced by grazing intensity and soil type on a shortgrass prairie site. J. Range Manage. 36:586-588.
Vinton, M.A., D.C. Hartnett, E.J. Finck and J.M. Briggs. 1993. Interactive effects of fire, bison *Bison bison* grazing and plant community composition in tallgrass prairie. Am. Midl. Nat. 129:10-18.
Vitousek, P.M., C.M. D'Antonio, L.L. Loope and R. Westbrooks. 1996. Biological invasions as global environmental change. Am. Sci. 84:468-478.
Vogel, W.G. 1987. A manual for training reclamation inspectors in the fundamentals of soils and revegetation. U.S. Dept. of Agric. Berea KY. 178 pp.
Vujnovic, K., R.W. Wein and M.R.T. Dale. 2002. Predicting plant species diversity in response to disturbance magnitude in grassland remnants of central Alberta. Can. J. Bot. 80:505-511.
Warren, S.D., M.B. Nevill, W.H. Blackburn and N.E. Garza. 1986. Soil response to trampling under intensive rotation grazing. Soil Sci. Soc. Am. J. 50:1336-1341.
Werner, S.J. and P.J. Urness. 1998. Elk forage utilization within rested units of rest-rotation grazing systems. J. Range Manage. 51:14-18.
Wright, R.G. 1999. Wildlife management in the national parks: questions in search of answers. Ecol. Appl. 9:30-36.
Wood, M.K. and W.H. Blackburn. 1981. Grazing systems: their influence on infiltration rates in the Rolling Plains of Texas. J. Range Manage. 34:331-335

Table 1. Native species revegetation seed mix at Elk Island National Park.

Common Name	Scientific Name	Mix Portion
Fringed brome	*Bromus ciliatus* L.	1/6
Green needle grass	*Stipa viridula* Trin.	1/6
Nodding brome	*Bromus anomalus* Rupr. Ex Fourn.	1/6
Slender wheatgrass	*Agropyron trachycaulum* (Link) Malte	1/6
Streambank wheatgrass	*Agropyron riparium* Scribn. and Smith	1/6
Western wheatgrass	*Agropyron smithii* Rydb.	1/6

Table 2. Mean daily temperature and monthly precipitation at Elk Island National Park meteorological station for 1998, 1999, 2000 and the long term average.

Mean Daily Temperature (degrees C)												
Year(s)	January	February	March	April	May	June	July	August	September	October	November	December
1971-2000	-13.3	-9.3	-4.1	4.4	10.9	14.5	16.6	16.1	10.6	4.6	-5.7	-10.5
1998	-16.8	-4.8	-3.8	7.5	15.3	15.9	19.3	18.8	12.3	6.1	-4.7	-11.1
1999	-14.3	-7.5	-3.8	5.1	9.1	13.7	15	17.3	10.6	5.3	-1.5	-3.9
2000	-13.3	-8.5	-3.1	3.5	8.8	13.3	17	14.9	10.2	4.4	-4.2	-14.6
Total Monthly Precipitation (mm)												
Year(s)	January	February	March	April	May	June	July	August	September	October	November	December
1971-2000	18.1	11.9	19.9	26	48.4	84.2	123.5	65.9	50.3	19.8	17.8	21
1997									84.4	25.1	7.6	7
1998	30.8	2.3	18.7	32.8	37	111.3	89.0	20.2	54.6	23.6	18.1	20
1999	48.1	4.2	8.2	16.4	66	22.2	89.4	45	16	6.8	18.9	13.3
2000	29.2	8.3	26.2	23.9	56	83.2	91.4	41.4				

Table 3. Mean soil particle size distribution with depth at Sites 1 and 2.

Site	Depth (cm)	Clay (%)	Silt (%)	Sand (%)	Texture
1	0-10	23.1 (1.6)	32.5 (6.0)	44.4 (7.3)	loam
2	0-10	23.5 (5.6)	35.0 (3.6)	41.6 (3.1)	loam
1	10-20	20.4 (1.3)	33.4 (8.1)	46.2 (7.2)	loam
2	10-20	26.4 (7.0)	38.0 (7.1)	35.6 (4.9)	loam
1	20-60	19.7 (1.3)	26.9 (5.3)	53.4 (6.)	sandy loam
2	20-60	39.5 (7.2)	31.6 (2.0)	28.9 (8.9)	clay loam
1	60-100	12.9 (1.9)	33.7 (9.5)	53.4 (9.1)	sandy loam
2	60-100	44.8 (8.8)	21.2 (4.4)	34.0 (5.2)	clay

Numbers in parentheses are standard errors

Table 4. Mean soil chemical properties with depth at Sites 1 and 2.

Site	Depth (cm)	pH	Electrical Conductivity ($dS\ m^{-1}$)	Total Carbon (%)	Organic Matter (%)
1	0-10	7.9 (0.2)	0.17 (0.01)	1.8 (0.1)	3.1
2	0-10	7.2 (0.2)	0.15 (0.03)	2.6 (1.3)	4.5
1	10-20	8.1 (0.1)	0.17 (0.01)	2.0 (0.1)	3.5
2	10-20	6.9 (0.3)	0.13 (0.03)	1.7 (0.4)	2.9
1	20-60	8.2 (0.1)	0.19 (0.01)	1.9 (0.3)	3.4
2	20-60	6.7 (0.6)	0.17 (0.09)	1.2 (0.2)	2.0
1	60-100	8.4 (0.1)	0.29 (0.03)	1.9 (0.1)	3.3
2	60-100	7.4 (0.5)	0.51 (0.17)	0.9 (0.1)	1.5

Numbers in parentheses are standard errors

Table 5. Mean plant density (plants m^{-2}) of seeded species for four treatment combinations in 1999 and 2000 at Elk Island National Park.

Seeded Species	Treatment Combinations and Year							
	Unfenced Broadcast		Unfenced Drilled		Fenced Broadcast		Fenced Drilled	
	1999	2000	1999	2000	1999	2000	1999	2000
Slender wheatgrass	18	106	10	47	27	69	17	42
Streambank wheatgrass	2	22	1	20	7	22	5	14
Western wheatgrass	5	48	6	61	2	2	5	15
Fringed brome	1	7	2	1	9	15	4	5
Nodding brome	0	0	0	0	0	0	0	0
Green needle grass	1	1	1	1	0	1	1	1
Total	27	184	20	130	45	109	32	77

Table 6. Mean plant density (plant m^{-2}) of non-seeded species for four treatment combinations in 1999 and 2000 at Elk Island National Park.

Non-Seeded Species	Treatment Combinations and Year							
	Unfenced Broadcast		Unfenced Drilled		Fenced Broadcast		Fenced Drilled	
	1999	2000	1999	2000	1999	2000	1999	2000
Alsike clover	6	52	3	120	11	13	14	119
American vetch	1	1	2	1	2	5	3	5
Balsam poplar	0	0	0	0	1	0	0	0
Biennial sagewort	0	0	1	0	0	0	0	0
Blue eyed grass	0	0	0	1	0	1	0	1
Canada anemone	0	0	0	0	1	1	0	1
Canada fleabane	0	0	0	0	1	0	1	0
Canada goldenrod	0	0	0	1	3	7	1	8
Canada thistle	9	7	4	13	5	7	4	8
Cinquefoil	2	0	3	0	4	0	1	0
Common chickweed	0	1	0	0	0	0	0	0
Common (stinging) nettle	0	0	2	0	1	2	0	2
Common peppergrass	1	0	1	0	0	0	0	0
Common wheat	0	0	1	0	0	0	0	0
Common wild rose	0	0	0	0	0	1	0	0
Cream colored vetchling	0	0	0	1	0	0	0	2
Curled dock	0	0	2	4	1	0	1	0
Dandelion	3	90	6	33	18	322	6	90
Dracocephalum spp.	0	0	1	0	0	0	0	0
Early blue violet	0	0	0	1	0	0	0	0
Fairy candelabra	0	0	0	1	0	0	0	0

Table 6. Mean plant density (plant m^{-2}) of non-seeded species for four treatment combinations in 1999 and 2000 at Elk Island National Park (continued).

Non-Seeded Species	Treatment Combinations and Year							
	Unfenced Broadcast		Unfenced Drilled		Fenced Broadcast		Fenced Drilled	
	1999	2000	1999	2000	1999	2000	1999	2000
Flixweed	2	0	1	0	0	0	0	0
Geranium	0	0	1	0	0	0	1	0
Giant hyssop	0	0	0	1	0	0	0	0
Grasses	4	0	9	0	2	0	2	0
Green tansy mustard	1	0	1	0	0	0	0	0
Hair grass	0	0	0	0	1	1	1	0
Hemp nettle	2	0	0	0	0	0	0	0
Horsetail	11	28	8	34	6	10	6	12
Kentucky bluegrass	0	51	0	46	4	42	2	20
Lamb's quarter	0	0	2	1	0	0	0	0
Macoun's wild rye	0	0	0	0	1	0	0	0
Mapleleaved goosefoot	0	1	0	0	0	0	0	0
Narrowleaved hawksbeard	3	0	1	2	0	1	1	1
Northern bedstraw	0	1	2	1	0	0	1	7
Northern willow herb	0	0	0	0	1	0	0	0
Perrenial sowthistle	0	7	0	5	0	7	1	22
Philadelphia fleabane	0	0	0	1	0	0	0	0
Plains wormwood	1	0	0	0	1	0	1	0
Plantain	6	9	8	16	13	8	8	4
Quack grass	0	0	0	1	0	0	0	0
Red clover	0	2	0	0	0	0	0	2

Table 6. Mean plant density (plant m^{-2}) of non-seeded species for four treatment combinations in 1999 and 2000 at Elk Island National Park (continued).

Non-Seeded Species	Treatment Combinations and Year							
	Unfenced Broadcast		Unfenced Drilled		Fenced Broadcast		Fenced Drilled	
	1999	2000	1999	2000	1999	2000	1999	2000
Red osier dogwood	0	0	0	0	0	2	0	0
Richard's ragwort	0	0	0	0	1	0	0	0
Sedges	0	0	0	0	0	0	1	1
Shepherd's purse	0	0	0	3	0	0	0	0
Smooth aster	0	0	0	0	0	1	0	0
Stinkweed	0	0	1	0	0	0	0	0
Strawberry blite	1	0	2	0	0	0	0	0
Striate knotweed	2	2	1	1	0	0	0	0
Weed seedlings	3	0	0	0	70	0	121	0
White clover	17	85	6	68	13	84	14	47
Wild strawberry	0	0	0	0	0	1	0	0
Yarrow	0	6	1	8	0	9	0	7
Yellow avens	0	0	1	0	0	1	0	0
Large leaved avens	0	0	0	1	0	0	0	1
Total Plants	75	343	72	367	161	526	191	360
Total Species	18	15	25	25	22	21	21	20

Table 7. Area utilized by wildlife in 1999 and 2000 at Sites 1 and 2 in Elk Island National Park.

Site 1 (196 m^2)						
	Unfenced Broadcast Seeded (96 m^2)		Unfenced Drill Seeded (96 m^2)			
	1999	2000	1999	2000		
Wildlife Indicators	Area (m^2)	Area (m^2)	Area (m^2)	Area (m^2)	1999 Total Area (m^2)	2000 Total Area (m^2)
Bedding	0	0	0	0	0	0
Grazing	54	51	52	39	106	90
Scat	5	5	7	2	12	7
Tracks	96	82	96	93	192	175
Wallows	0	0	0	0	0	0
Total Area Utilized	155	138	155	134	310	272
Site 2 (410 m^2)						
	Unfenced Broadcast Seeded (205 m^2)		Unfenced Drill Seeded (205 m^2)			
	1999	2000	1999	2000		
Wildlife Indicators	Area (m^2)	Area (m^2)	Area (m^2)	Area (m^2)	1999 Total Area (m^2)	2000 Total Area (m^2)
Bedding	3	19	0	12	3	31
Grazing	93	54	74	1	167	55
Scat	27	7	30	17	57	24
Tracks	205	83	204	119	409	202
Wallows	1	0	27	0	28	0
Total Area Utilized	329	163	335	149	664	312

Total area utilized at both sites in 1999 is 974 m^2 and in 2000 is 584 m^2
Total area utilized in all treatment combinations for 1999 and 2000 is 1558 m^2
Site 1 total area is 192 m^2, Site 2 total area is 410 m^2

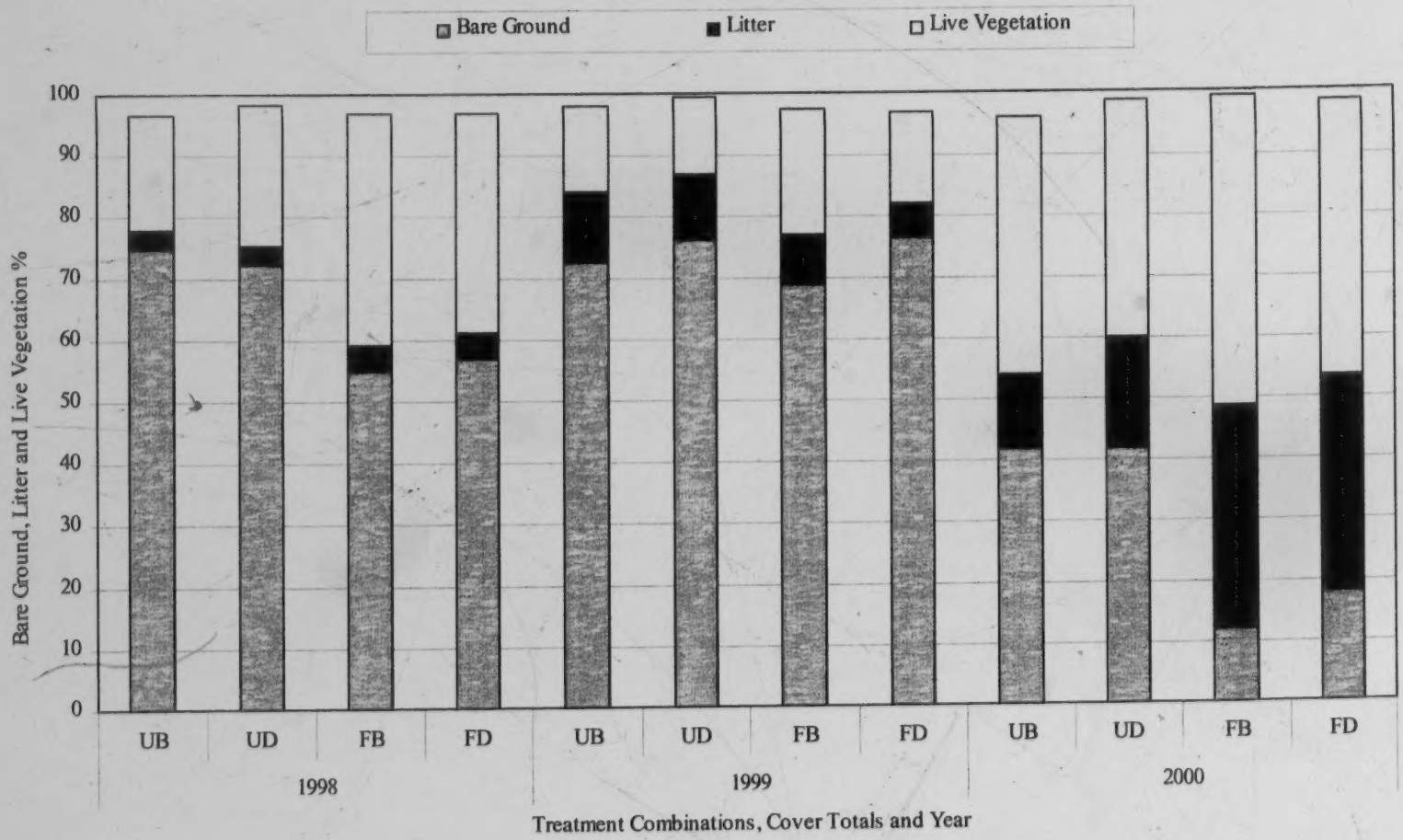


Figure 1. Mean bare ground, litter and live vegetation for four treatment combinations in 1998, 1999 and 2000 at Elk Island National Park. Where UB=unfenced broadcast seeded, UD=unfenced drill seeded, FB=fenced broadcast seeded and FD=fenced drill seeded

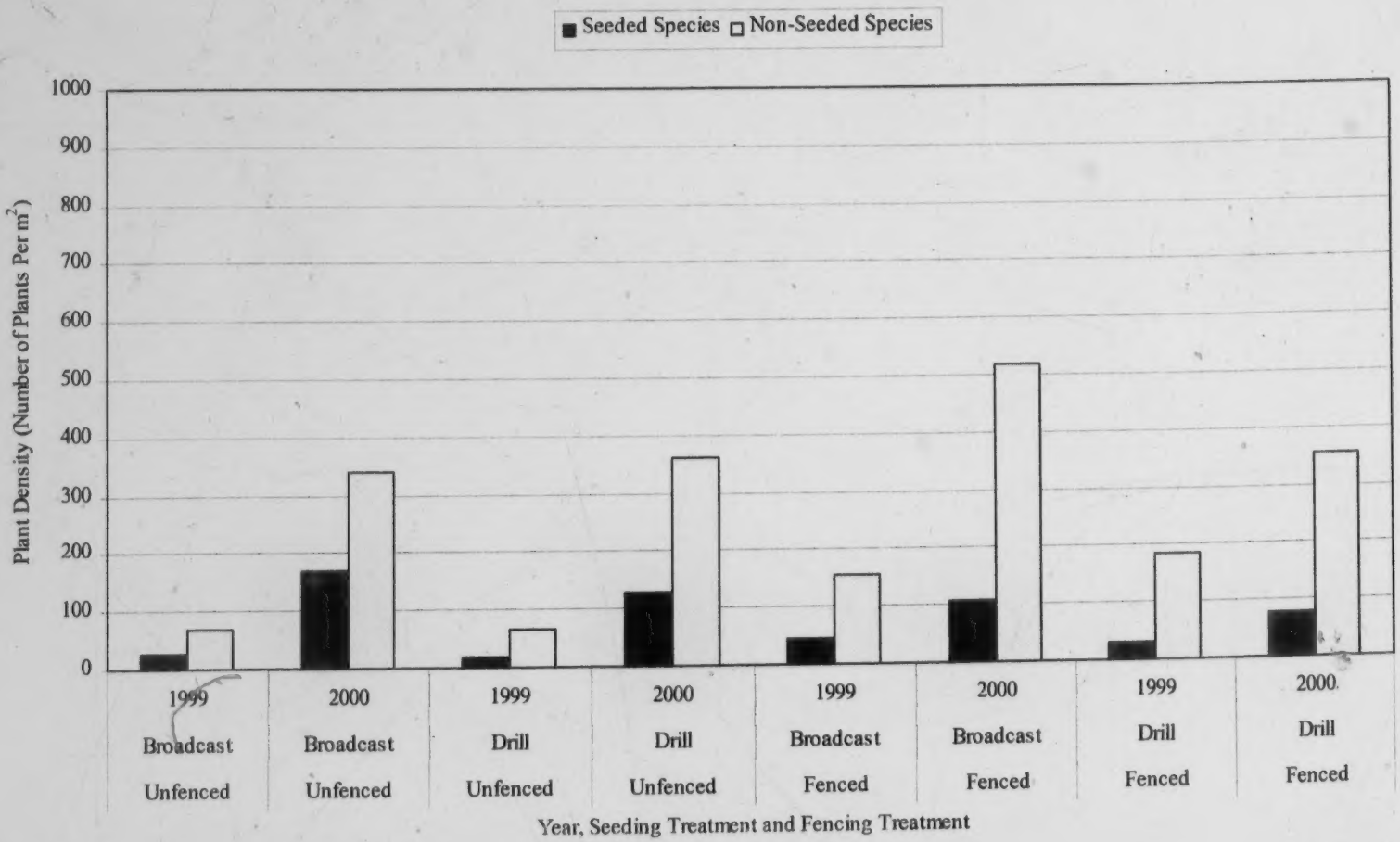


Figure 2. Mean plant density of seeded and non-seeded species for four treatment combinations in 1999 and 2000 at Elk Island National Park.

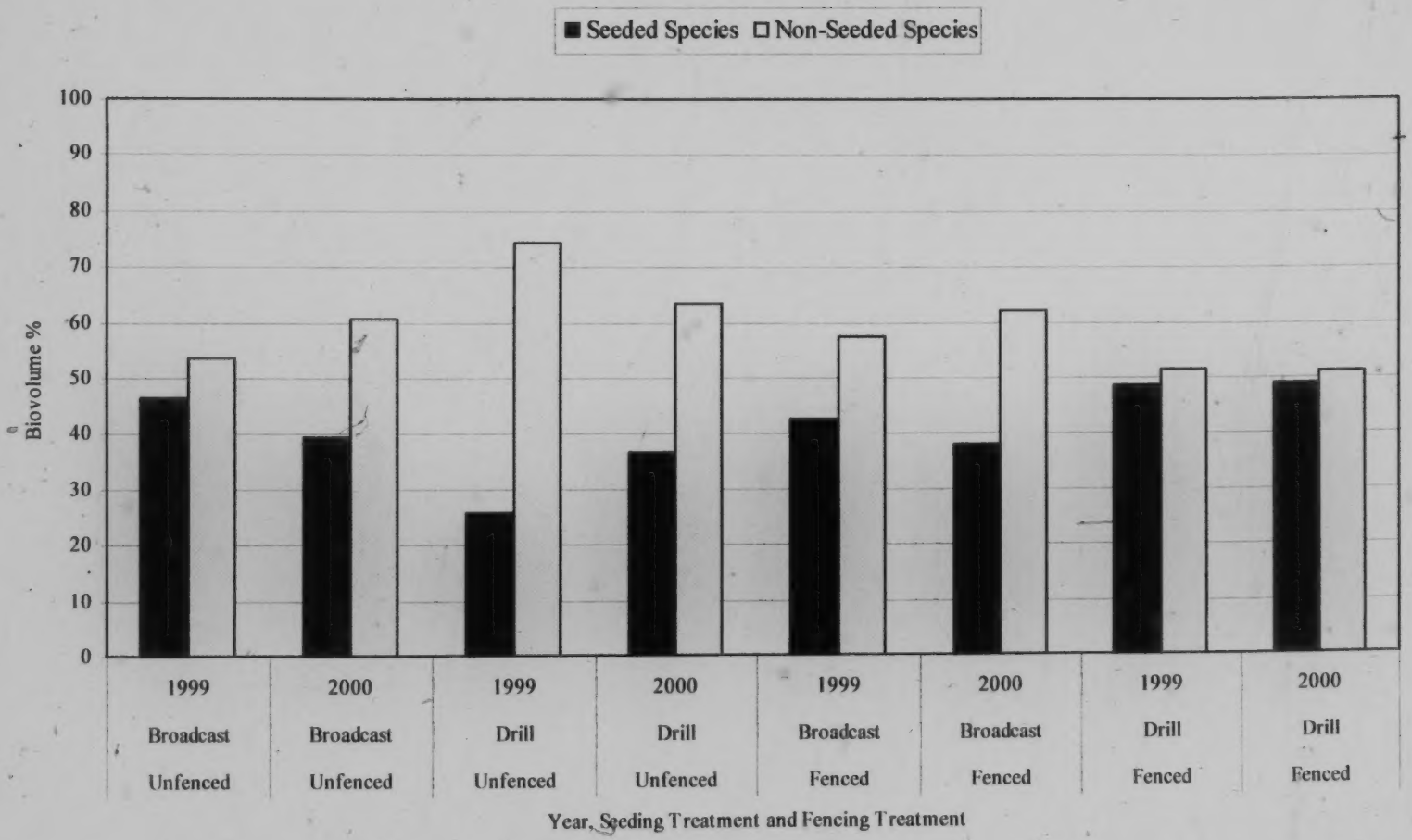


Figure 3. Mean plant biovolume of seeded and non-seeded species for four treatment combinations in 1999 and 2000 at Elk Island National Park.

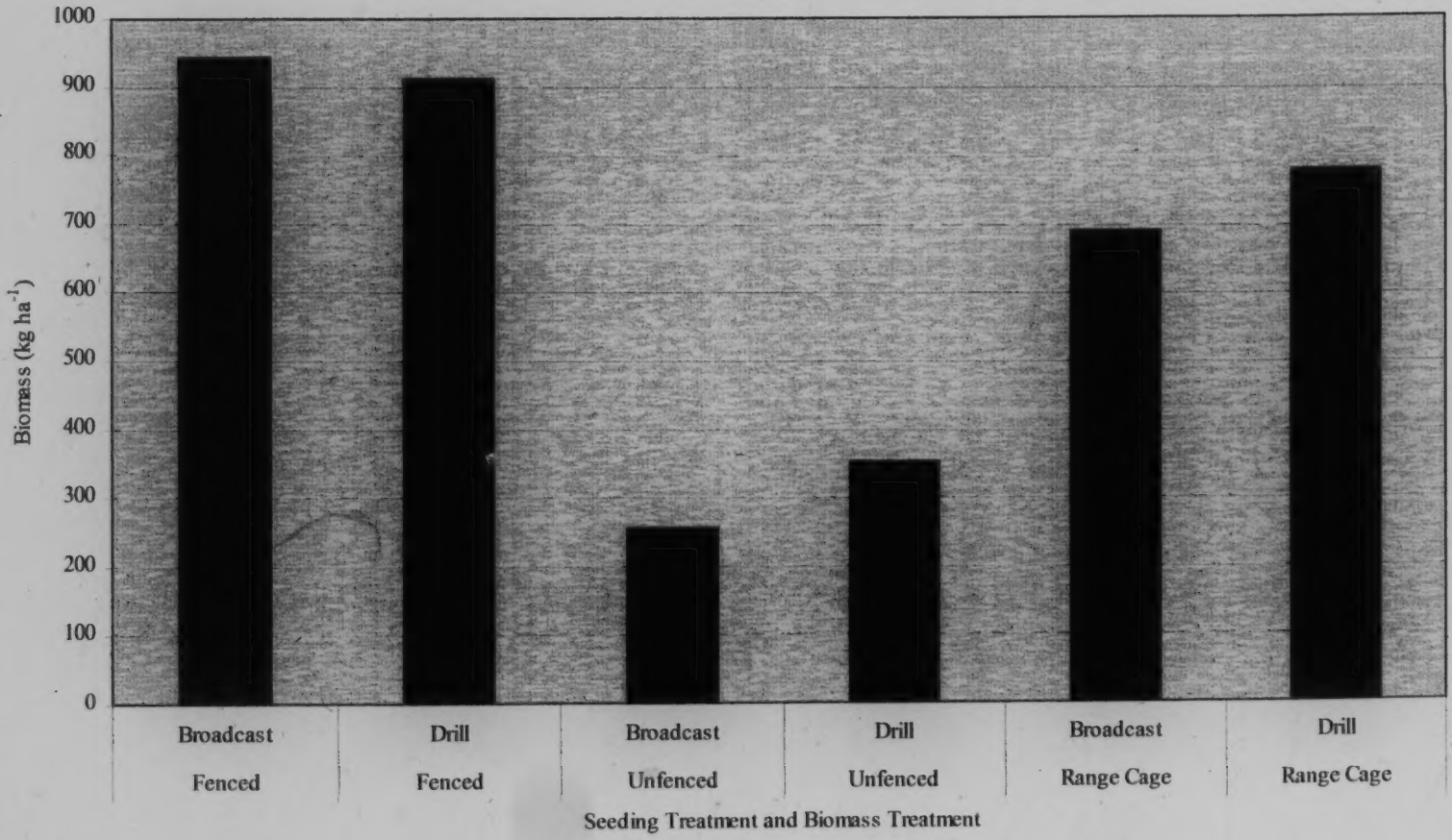


Figure 4. Mean plant biomass of seeded and non-seeded species for six treatment combinations in 2000 at Elk Island National Park.

APPENDIX

Table A1. Petroleum parameters and polychlorinated biphenyls (PCB) (mg kg^{-1}) from soil analyses at Sites 1 and 2 in Elk Island National Park.

Parameter	Site 1		Site 2			Detection Limit
	Sample 1	Sample 2	Sample 1	Sample 2	Sample 3	
	3.7 m	3.0 m	1.2 m	2.4 m	1.2 m	
Flashpoint (°C)	NA	NA	NA	NA	39.0	0.1
Benzene	ND	ND	ND	ND	ND	NS
Toluene	ND	ND	ND	ND	ND	0.01
Ethylbenzene	ND	ND	ND	ND	ND	0.01
Xylenes	ND	ND	ND	ND	ND	0.01
PHC F1 (C_6-C_{10})	ND	ND	ND	ND	ND	0.01
PHC F2 (C_{10}-C_{16})	ND	ND	ND	6	ND	5
PHC F3 (C_{16}-C_{34})	70	160	150	260	98	5
PHC F4 (C_{34}-C_{50})	300	800	110	1100	1400	100
PCBs	NA	ND	NA	NA	ND	0.05

Adapted from O'Connor Associates (2001)

ND=Not detected
NA=Not analyzed

Table A2. Volatile organic compound (mg kg^{-1}) soil analyses at Sites 1 and 2 in Elk Island National Park.

Parameter	Site 1		Site 2			Detection Limit
	Sample 1 3.7 m	Sample 2 3.0 m	Sample 1 1.2 m	Sample 2 2.4 m	Sample 3 1.2 m	
Acetone	ND	ND	ND	ND	ND	1
Acrolein	ND	ND	ND	ND	ND	1
Acrylonitrile	ND	ND	ND	ND	ND	1
Benzene	ND	ND	ND	ND	ND	0.01
Bromodichloromethane	ND	ND	ND	ND	ND	0.01
Bromoform	ND	ND	ND	ND	ND	0.03
Bromomethane	ND	ND	ND	ND	ND	0.1
2-Butanone (MEK)	ND	ND	ND	ND	ND	1
Carbon disulfide	ND	ND	ND	ND	ND	0.01
Carbon tetrachloride*	ND	ND	ND	ND	ND	0.01
Chlorobenzene	ND	ND	ND	ND	ND	0.01
Chloroethane	ND	ND	ND	ND	ND	0.1
2-Chloroethyl vinyl ether	ND	ND	ND	ND	ND	0.1
Chloroform*	ND	ND	ND	ND	ND	0.01
Chloromethane	ND	ND	ND	ND	ND	0.1
Dibromochloromethane	ND	ND	ND	ND	ND	0.03
1,2-Dibromoethane	ND	ND	ND	ND	ND	0.01
Dibromomethane	ND	ND	ND	ND	ND	0.03
1,2-Dichlorobenzene	ND	ND	ND	ND	ND	0.01
1,3-Dichlorobenzene	ND	ND	ND	ND	ND	0.01
1,4-Dichlorobenzene	ND	ND	ND	ND	ND	0.01
cis-1,4-Dichloro-2-butene	ND	ND	ND	ND	ND	0.1
trans-1,4-Dichloro-2-butene	ND	ND	ND	ND	ND	0.1
Dichlorodifluoromethane	ND	ND	ND	ND	ND	0.03
1,1-Dichloroethane*	ND	ND	ND	ND	ND	0.01
1,2-Dichloroethane*	ND	ND	ND	ND	ND	0.02
1,1-Dichloroethene*	ND	ND	ND	ND	ND	0.01
trans-1,2-Dichloroethene*	ND	ND	ND	ND	ND	0.01
1,2-Dichloropropane*	ND	ND	ND	ND	ND	0.02
cis-1,3-Dichloropropene*	ND	ND	ND	ND	ND	0.01

Adapted from O'Connor Associates (2001)

ND=Not detected

Table A2. Volatile organic compound (mg kg^{-1}) soil analyses at Sites 1 and 2 in Elk Island National Park (continued).

Parameter	Site 1		Site 2			Detection Limit
	Sample 1 3.7 m	Sample 2 3.0 m	Sample 1 1.2 m	Sample 2 2.4 m	Sample 3 1.2 m	
trans-1,3-Dichloropropene*	ND	ND	ND	ND	ND	0.01
Ethanol	ND	ND	ND	ND	ND	3
Ethylbenzene	ND	ND	ND	ND	ND	0.01
Ethyl methacrylate	ND	ND	ND	ND	ND	0.1
2-Hexanone	ND	ND	ND	ND	ND	0.1
Iodomethane	ND	ND	ND	ND	ND	0.01
4-Methyl-2-pentanone (MIBK)	ND	ND	ND	ND	ND	0.1
Methylene chloride	ND	ND	0.05	0.06	0.02	0.01
Styrene	ND	ND	ND	ND	ND	0.01
Tetrachloroethylene*	ND	ND	ND	ND	ND	0.01
1,1,2,2-Tetrachloroethane*	ND	ND	ND	ND	ND	0.2
1,1,1-Trichloroethane*	ND	ND	ND	ND	ND	0.01
1,1,2-Trichloroethane*	ND	ND	ND	ND	ND	0.02
1,2,3-Trichloropropane	ND	ND	ND	ND	ND	0.05
Trichloroethene	ND	ND	ND	ND	ND	0.01
Trichlorofluoromethane	ND	ND	ND	ND	ND	0.01
Toluene	ND	ND	ND	ND	ND	0.01
Vinyl acetate	ND	ND	ND	ND	ND	1
Vinyl chloride	ND	ND	ND	ND	ND	0.02
Xylenes (m,p,o)	ND	ND	ND	ND	ND	0.02
Sum of * halogenated aliphatics	ND	ND	ND	ND	ND	0.35
Sum of chlorobenzenes	ND	ND	ND	ND	ND	0.04

Adapted from O'Connor Associates (2001)

ND=Not detected

Table A3. Polycyclic aromatic hydrocarbon (mg kg^{-1}) soil analyses at Sites 1 and 2 in Elk Island National Park.

Parameter	Site 1		Site 2			Detection Limit
	Sample 1 3.7 m	Sample 2 3.0 m	Sample 1 1.2 m	Sample 2 2.4 m	Sample 3 1.2 m	
Naphthalene	ND	ND	ND	0.01	ND	0.01
Acenaphthylene	ND	ND	ND	ND	ND	0.01
Acenaphthene	ND	ND	ND	ND	ND	0.01
Anthracene	ND	ND	ND	ND	ND	0.01
Fluorene	ND	ND	ND	ND	ND	0.01
Phenanthrene	ND	ND	ND	ND	ND	0.01
Fluoranthene	ND	ND	ND	ND	ND	0.01
Pyrene	ND	ND	ND	ND	ND	0.01
B(a)A/Chrysene/B(c)P[a]	ND	ND	ND	0.01	ND	0.01
Benzo (b,j,k) fluoranthene	ND	ND	ND	ND	ND	0.01
Benzo(a)pyrene	ND	ND	ND	ND	ND	0.01
3-Methylchloranthene	ND	ND	ND	ND	ND	0.01
Indeno(1,2,3-cd)pyrene	ND	ND	ND	ND	ND	0.01
Dibenzo(a,h)anthracene	ND	ND	ND	ND	ND	0.01
7,12-Dimethylbenz(a)anthracene	ND	ND	ND	ND	ND	0.01
Benzo(g,h,i)perylene	ND	ND	ND	ND	ND	0.01
Dibenzo(a,h/a,i/a,j)pyrene	ND	ND	ND	ND	ND	0.01

Adapted from O'Connor Associates (2001)

[a] sum of benzo(a) anthracene, chrysene and benzo(c) phenanthrene
ND=Not detected

Table A4. Seeded native grass species identified in Sites 1 and 2 at Elk Island National Park.

Common Name	Scientific Name	Life History	Habitat
Fringed brome	*Bromus ciliatus* L.	Perennial	Meadows, open woods
Green needle grass	*Stipa viridula* Trin.	Perennial	Dry prairie grassland
Nodding brome	*Bromus anomalus* Rupr. *Ex* Fourn.	Perennial	Grassland, moist prairie, open woods
Slender wheatgrass	*Agropyron trachycaulum* (Link) Malte	Perennial	Open woods, parkland, prairie
Streambank wheatgrass	*Agropyron riparium* Scribn. and Smith	Perennial	Dry open woodland, grassland prairie
Western wheatgrass	*Agropyron smithii* Rydb.	Perennial	Grassland, prairie

Adapted from Hardy BBT Limited (1989), Moss (1996) and Royer and Dickinson (1999)

Table A5. Non-seeded plant species identified at Sites 1 and 2 at Elk Island National Park.

Common Name	Scientific Name	Life History	Habitat
Alsike clover	*Trifolium hybridium* L.	Perennial	Fields, waste ground
American vetch	*Vicia americana* Muhl.	Perennial	Meadows, open woods
Balsam poplar	*Populus balsamifera* L.	Perennial	Boreal forest,
Blue eyed grass	*Sisyrinchium montanum* Greene	Perennial	Moist open areas
Biennial sagewort	*Artemisia biennis* Willd.	Annual, biennial	Disturbed areas, moist ground
Canada anemone	*Anemone canadensis* L.	Perennial	Damp meadows
Canada fleabane	*Erigeron canadensis* L.	Annual	Disturbed ground, fields
Canada goldenrod	*Solidago canadensis* L.	Perennial	Moist meadows, open woods
Canada thistle	*Cirsium arvense* (L.) Scop.	Perennial	Fields, waste ground
Cinquefoil	*Potentilla norvegica* L.	Annual, biennial	Waste ground, wet meadows
Common chickweed	*Stellaria media* (L.) Cyrill.	Annual	Waste ground
Common stinging nettle	*Urtica dioica* L.	Perennial	Moist shady woodland
Common peppergrass	*Lepidium densiflorum* Schrad.	Annual, winter annual	Disturbed habitats, dry open areas
Common wheat	*Triticum aestivum* L.	Annual	An escape from cultivation
Common wild rose	*Rosa woodsii* Lindl.	Perennial	Open woods
Cream colored vetchling	*Lathyrus ochroleucus* Hook.	Perennial	Clearings, moist woods
Curled dock	*Rumex crispus* L.	Perennial	Moist ground, waste places
Dandelion	*Taraxacum officinale* Weber	Perennial	Waste ground
Dragonhead spp.	*Dracocephalum* spp.	Annual, biennial, perennial	Disturbed areas, open woods, waste ground
Dragonwort	*Artemisia dracunculus* L.	Perennial	Dry hills, prairies
Early blue violet	*Viola adunca* J.E. Smith	Perennial	Grassland, open woods
Fairy candelabra	*Androsace septentrionalis* L.	Winter annual, short lived perennial	Dry open ground
Flixweed	*Descurainia sophia* (L.) Webb.	Annual, biennial	Fields, waste ground

Table A5. Non-seeded plant species identified at Sites 1 and 2 at Elk Island National Park (continued).

Common Name	Scientific Name	Life History	Habitat
Fowl bluegrass	*Poa palustris* L.	Perennial	Moist meadows, open woods
Geranium	*Geranium* spp.	Annual, biennial	Clearings, disturbed soils, open woods
Grasses	*Poa* spp.		
Green tansy mustard	*Descurainia pinnata* (Walt.) Britt.	Annual, biennial	Dry open areas, waste ground
Giant hyssop	*Agastache foeniculum* (Pursh) Ktze.	Perennial	Grasslands
Hair grass	*Agrostis scabra* Willd.	Perennial	Disturbed areas, open woods
Hemp nettle	*Galeopsis tetrahit* L.	Annual	Fields waste ground
Horsetail	*Equisetum arvense* L.	Perennial	Meadows, moist woods
Kentucky bluegrass	*Poa pratensis* L.	Perennial	Prairies, woodland
Lamb's quarter	*Chenopodium album* L.	Annual	Waste ground
Macoun's wild rye	*Agrohordeum macounii* (Vasey) Lepage	Perennial	Moist meadows, saline flats
Maple leaved goosefoot	*Chenopodium gigantospermum* Aellen	Annual	Dry to moist, waste ground, woodlands
Narrow leaved hawksbeard	*Crepis tectorum* L.	Annual	Waste ground
Northern bedstraw	*Galium boreale* L.	Annual	Moist prairie valleys, woodlands
Northern willow herb	*Epilobium ciliatum* Raf.	Perennial	Disturbed moist areas
Perrenial sowthistle	*Sonchus arvense* L.	Perennial	Waste ground
Philadelphia fleabane	*Erigeron philadelphicus* L.	Annual	Clearings, disturbed ground
Pineapple weed	*Matricaria matricarioides* (Less.) Porter	Annual	Waste ground
Plantain	*Plantago major* L.	Perennial	Waste ground
Plains muhly	*Muhlenbergia cuspidata* (Torr.) Rydb.	Perennial	Dry slopes

Table A5. Non-seeded plant species identified at Sites 1 and 2 at Elk Island National Park (continued).

Common Name	Scientific Name	Life History	Habitat
Plains wormwood	*Artemisia campestris* L.	Biennial, perennial	Prairies
Prickly rose	*Rosa acicularis* Lindl.	Perennial	Banks, woods
Quack grass	*Agropyron repens* (L.) Beauv.	Perennial	Waste ground
Red clover	*Trifolium pratense* L.	Biennial, short-lived perennial	Fields, waste ground
Red osier dogwood	*Cornus stolonifera* Michx.	Perennial	Moist woods
Richard's ragwort	*Senecio eremophilus* Richards	Perennial	Moist open woods
Sagebrush	*Artemisia cana* Pursh	Perennial	Dry hills, prairies
Sedges	*Carex* spp.	Perennial	
Shepherd's purse	*Capsella bursa-pastoris* (L.) Medic.	Annual	Fields
Smooth aster	*Aster laevis* L.	Perennial	Open dry areas
Smooth brome	*Bromus inermis* Leyss.	Perennial	Fields, grasslands, open woods
Snowberry	*Symphoricarpos albus* (L.) Blake	Perennial	Open woodland
Stinkweed	*Thlapsi arvense* L.	Annual	Fields, waste ground
Strawberry blite	*Chenopodium capitatum* (L.) Aschers.	Annual	Clearings, waste ground
Striate knotweed	*Polygonum erectum* L.	Annual	Waste places
Western willow aster	*Aster hesperius* A. Gray	Perennial	Moist ground, stream banks
White clover	*Trifolium repens* L.	Perennial	Fields
Wild red raspberry	*Rubus idaeus* L.	Perennial	Open woods
Wild strawberry	*Fragaria virginiana* Duchesne	Perennial	Plains woods
Yellow avens	*Geum aleppicum* Jacq.	Perennial	Meadows, moist woods
Large leaved avens	*Geum macrophyllum* Willd.	Perennial	Meadows, moist woods
Yarrow	*Achillea millefolium* L.	Perennial	Clearings, prairie, waste ground

Adapted from Hardy BBT Limited (1989), Moss (1996) and Royer and Dickinson (1999)

Table A6. Non-seeded, native species identified in Sites 1 and 2 at Elk Island National Park.

Common Names	Scientific Names
Forbs	
American vetch	*Vicia americana* Muhl.
Biennial sagewort	*Artemisia biennis* Willd.
Blue eyed grass	*Sisyrinchium montanum* Greene
Canada anemone	*Anemone canadensis* L.
Canada fleabane	*Erigeron canadensis* L.
Canada goldenrod	*Solidago canadensis* L.
Cream colored vetchling	*Lathyrus ochroleucus* Hook.
Common (stinging) nettle	*Urtica dioica* L.
Common peppergrass	*Lepidium densiflorum* Schrad.
Dragonhead spp.	*Dracocephalum spp.*
Dragonwort	*Artemisia dracunculus* L.
Early blue violet	*Viola adunca* J.E. Smith
Fairy candelabra	*Androsace septentrionalis* L.
Geranium	*Geranium* spp.
Green tansy mustard	*Descurainia pinnata* (Walt.) Britt.
Giant hyssop	*Agastache foeniculum* (Pursh) Ktze.
Horsetail	*Equisetum arvense* L.
Mapleleaved goosefoot	*Chenopodium gigantospermum* Aellen
Northern bedstraw	*Galium boreale* L.
Northern willow herb	*Epilobium ciliatum* L.
Philadelphia fleabane	*Erigeron philadelphicus* L.
Plains wormwood	*Artemisia campestris* L.
Richard's ragwort	*Senecio eremophilus* Richards
Smooth aster	*Aster laevis* L.
Strawberry blite	*Chenopodium capitatum* (L.) Aschers.
Striate knotweed	*Polygonum erectum* L.
Western willow aster	*Aster hesperius* A. Gray
Wild strawberry	*Fragaria virginiana* Duchesne
Yellow avens	*Geum aleppicum* Jacq.
Large leaved avens	*Geum macrophyllum* Willd.
Yarrow	*Achillea millefolium* L.
Grasses	
Fowl bluegrass	*Poa palustris* L.
Hair grass	*Agrostis scabra* Willd.
Macoun's wild rye	*Agrohordeum macounii* (Vasey) Lepage
Plains muhly	*Muhlenbergia cuspidata* (Torr.) Rydb.

Table A6. Non-seeded, native species identified in Sites 1 and 2 at Elk Island National Park (continued).

Common Names	Scientific Names
Shrubs and Trees	
Balsam poplar	*Populus balsamifera* L.
Cinquefoil	*Potentilla norvegica* L.
Common wild rose	*Rosa woodsii* Lindl.
Prickly rose	*Rosa acicularis* Lindl.
Red osier dogwood	*Cornus stolonifera* Michx.
Sagebrush	*Artemisia cana* Pursh
Snowberry	*Symphoricarpos albus* (L.) Blake
Wild red raspberry	*Rubus idaeus* L.

Adapted from Hardy BBT Limited (1989), Moss (1996) and Royer and Dickinson (1999)

Table A7. Non-seeded, non-native species identified in Sites 1 and 2 at Elk Island National Park.

Common Names	Scientific Names
Forbs	
Alsike clover	*Trifolium hybridium* L.
Canada thistle	*Cirsium arvense* (L.) Scop.
Common chickweed	*Stellaria media* (L.) Cyrill.
Curled dock	*Rumex crispus* L.
Dandelion	*Taraxacum officinale* Weber
Flixweed	*Descurainia sophia* (L.) Webb.
Hemp nettle	*Galeopsis tetrahit* L.
Lamb's quarter	*Chenopodium album* L.
Narrowleaved hawksbeard	*Crepis tectorum* L.
Perrenial sowthistle	*Sonchus arvense* L.
Pineapple weed	*Matricaria matricarioides* (Less.) Porter
Plantain	*Plantago major* L.
Red clover	*Trifolium pratense* L.
Shepherd's purse	*Capsella bursa-pastoris* (L.) Medic.
Stinkweed	*Thlapsi arvense* L.
White clover	*Trifolium repens* L.
Grasses	
Common wheat	*Triticum aestivum* L.
Kentucky bluegrass	*Poa pratensis* L.
Quack grass	*Agropyron repens* (L.) Beauv.
Smooth brome	*Bromus inermis* Leyss.

Adapted from Hardy BBT Limited (1989), Moss (1996) and Royer and Dickinson (1999)

Table A8. Statistical linear model for the split plot design.

$$Y_{ijk} = \mu + \rho_i + \alpha_j + \rho\alpha_{ij} + \beta_k + (\alpha\beta)_{jk} + \varepsilon_{ijk}$$

where i = site
j = fencing treatment
k = seeding treatment

and

Y_{ijk} =	the average yield over n observations of the k^{th} seeding treatment in the j^{th} fencing treatment in the i^{th} site
μ =	the overall mean
ρ_i =	the block effect associated with the i^{th} block (random effect)
α_j =	the main plot treatment effect (factor A) associated with the j^{th} fencing treatment (fixed effect)
$\rho\alpha_{ij}$ =	the main plot error, which is the interaction between site and fencing treatment, referred to as error (a)
β_k =	the subplot treatment effect (factor B) associated with the k^{th} seeding treatment (fixed effect)
$(\alpha\beta)_{jk}$ =	the interaction effect associated with the j^{th} fencing treatment and the kth seeding treatment
ε_{ijk} =	the residual error, referred to as error (b)

Table A9. Analysis of variance table, split plot design with block as a random effect and factors A and B as fixed effects.

Source of Variation	Degrees of Freedom Calculations	Degrees of Freedom
Block (Site)	(r-1) = (2-1)	1
Main plot factor [Fence (Factor A)]	(a-1) = (2-1)	1
Main plot error [Error (a)]	(r-1)(a-1) = (2-1)(2-1)	1
Subplot factor [Seed method (Factor B)]	(b-1) = (2-1)	1
Interaction [Fence x Seed Method (Factor A x Factor B)]	(a-1)(b-1) = (2-1)(2-1)	1
Experimental error [Error (b)]	a(r-1)(b-1) = 2(2-1)(2-1)	2
Total	(rab-1) = [(2)(2)(2)-1]	7
Main plot total	(ar-1) = [(2)(2)-1]	3
Subplot total	ar(b-1) = [(2)(2)(2-1)]	4
Total		7

Table A10. Statistical linear model for the split plot design when error (a) is pooled with error (b).

$$Y_{ijk} = \mu + \rho_i + \alpha_j + \beta_k + (\alpha\beta)_{jk} + \varepsilon_{ijk}$$

where i = site
j = fencing treatment
k = seeding treatment

and

Y_{ijk} = the average yield over n observations of the k^{th} seeding treatment in the j^{th} fencing treatment in the i^{th} site

μ = the overall mean

ρ_i = the block effect associated with the i^{th} block (random effect)

α_j = the main plot treatment effect (factor A) associated with the j^{th} fencing treatment (fixed effect)

β_k = the subplot treatment effect (factor B) associated with the k^{th} seeding treatment (fixed effect)

$(\alpha\beta)_{jk}$ = the interaction effect associated with the j^{th} fencing treatment and the kth seeding treatment

ε_{ijk} = the residual error

Table A11. Analysis of variance table for a split plot design with block as a random effect and factors A and B as fixed effects when error (a) is pooled with error (b).

Source of Variation	Degrees of Freedom Calculations	Degrees of Freedom
Block (Site)	(r-1) = (2-1)	1
Main plot factor [Fence (Factor A)]	(a-1) = (2-1)	1
Subplot factor [Seed method (Factor B)]	(b-1) = (2-1)	1
Interaction [Fence x Seed Method (Factor A x Factor B)]	(a-1)(b-1) = (2-1)(2-1)	1
Experimental error	a(r-1)(b-1) = 2(2-1)(2-1) + 1(error a)	3
Total	(rab-1) = [(2)(2)(2)-1]	7
Main plot total	(ar-1) = [(2)(2)-1]	2
Subplot total	ar(b-1) = [(2)(2)(2-1)] + 1(error a)	5
Total		7

Table A12. Plant reclamation suitability criteria for seeded species.

Scientific Name	Growth Form	Reproduction	Ecology Physiology	Forage Value	Grazing Response	Palatability	Browse Tolerance
Agropyron riparium	0.4 to 1.3 m erect bunch, sod	Rhizomes, seeds, tillers	Early and late seral stages Cool growing season	Good for wildlife	Increaser/decreaser	Low	High
Agropyron smithii	0.3 to 0.6 m erect bunch, sod	Rhizomes, seeds, tillers	Early and late seral stages Cool growing season	Fair for wildlife	Increaser/decreaser	Medium to high	Medium
Agropyron trachycaulum	0.5 to 1.5 m erect bunch	Rhizomes, seeds, tillers	Late seral stage Cool growing season	Good for wildlife	Increaser/decreaser	Medium	Low
Bromus anomalus	0.3 to 0.6 m erect bunch	Seeds	Early seral stage		Increaser/decreaser		
Bromus ciliatus	0.6 to 1.0 m erect bunch	Seeds	Early and late seral stages Cool growing season	Excellent for wildlife	Decreaser		Low
Stipa viridula	0.4 to 0.8 m erect bunch	Seeds, tillers	Early seral stage, Cool growing season	Good for wildlife	Increaser/decreaser	Medium to high	Low to medium

Table A12. Plant reclamation suitability criteria for seeded species (continued).

Scientific Name	Winter Hardiness	Drought Tolerance	Persistance	Salt Tolerance	pH Tolerance	Soil Preference	Erosion Control	Moisture Preference
Agropyron riparium	Medium to high	High	Very high	Medium	6.0 to 9.5	Well drained soils, tolerates medium to coarse textural range	Excellent	Moist to dry, withstands spring flooding
Agropyron smithii	Medium	Medium to high	Very high	Medium to high	5.0	Clay and clay-loamed, fine to medium textural range	Excellent	Wet to dry, withstands spring flooding
Agropyron trachycaulum	High	Medium	Medium	Medium to high	8.8	Well drained, medium textural range	Excellent	Moist to dry, withstands spring flooding
Bromus anomalus								Wet to dry
Bromus ciliatus								Moist
Stipa viridula	Medium	Medium	Medium	Low		Loamy to clay, fine to medium textured	Medium	Moist to dry

Adapted from Hardy BBT Limited (1989) and Gerling et al. (1996)

Table A13. Mean bare ground, litter, live vegetation and combined totals for four treatment combinations in 1998, 1999 and 2000 at Sites 1 and 2 in Elk Island National Park.

	Bare Ground			Litter			Live Vegetation		
Treatment	Site 1	Site 2	Mean	Site 1	Site 2	Mean	Site 1	Site 2	Mean
1998									
UB	86.4	63.1	74.8 (11.68)	2.4	3.3	2.9 (0.45)	6.2	31.4	18.8 (12.64)
UD	74.3	69.8	72.1 (2.26)	3.9	2.3	3.1 (0.8)	20	25.9	23 (2.96)
FB	61.3	48.4	54.9 (6.47)	3	5.6	4.3 (1.3)	30.9	44.6	37.8 (6.87)
FD	56.6	57.3	57 (0.35)	2.6	6	4.3 (1.71)	37.4	34.3	35.9 (1.55)
1999									
UB	87.8	57	72.4 (15.45)	1.2	22	11.6 (10.43)	7.7	20.3	14 (6.32)
UD	79.5	72.4	76 (3.56)	1.2	20.3	10.8 (9.58)	18	7.3	12.7 (5.37)
FB	66.7	70.8	68.8 (2.06)	8.2	8.1	8.2 (0.05)	20.2	20.5	20.4 (0.15)
FD	73.9	78.6	76.3 (2.36)	5.6	5.7	5.7 (0.05)	14.5	15.1	14.8 (0.3)
2000									
UB	43.7	40.1	41.9 (1.81)	10.6	13.3	12 (1.35)	44.4	39.8	42.1 (2.31)
UD	41.4	42.2	41.8 (0.40)	12.6	23.5	18.1 (5.47)	43.7	33.4	38.6 (5.17)
FB	16.8	6.8	11.8 (5.01)	32.5	40.9	36.7 (4.21)	49.1	52.2	50.7 (1.55)
FD	13.2	22.6	17.9 (4.71)	42.4	28.5	35.5 (6.97)	43.4	46.5	45 (1.55)

Where, UB=unfenced broadcast seeded, UD=unfenced drill seeded, FB=fenced broadcast seeded and FD=fenced drill seeded
Numbers in parentheses are standard errors

Table A14. Dependent variables and associated p-values for treatment interactions and main effects in 1998 at Elk Island National Park.

Dependent Variable	Treatment Interaction	Fencing Treatment	Seeding Treatment
Bare Ground	0.6986	0.0349*	0.9639
Litter	0.8421	0.2486	0.8421
Live Vegetation	0.6798	0.0612*	0.8746

Where significance is set at p-values of 0.10

Table A15. Dependent variables and associated p-values for treatment interactions and main effects in 1999 at Elk Island National Park.

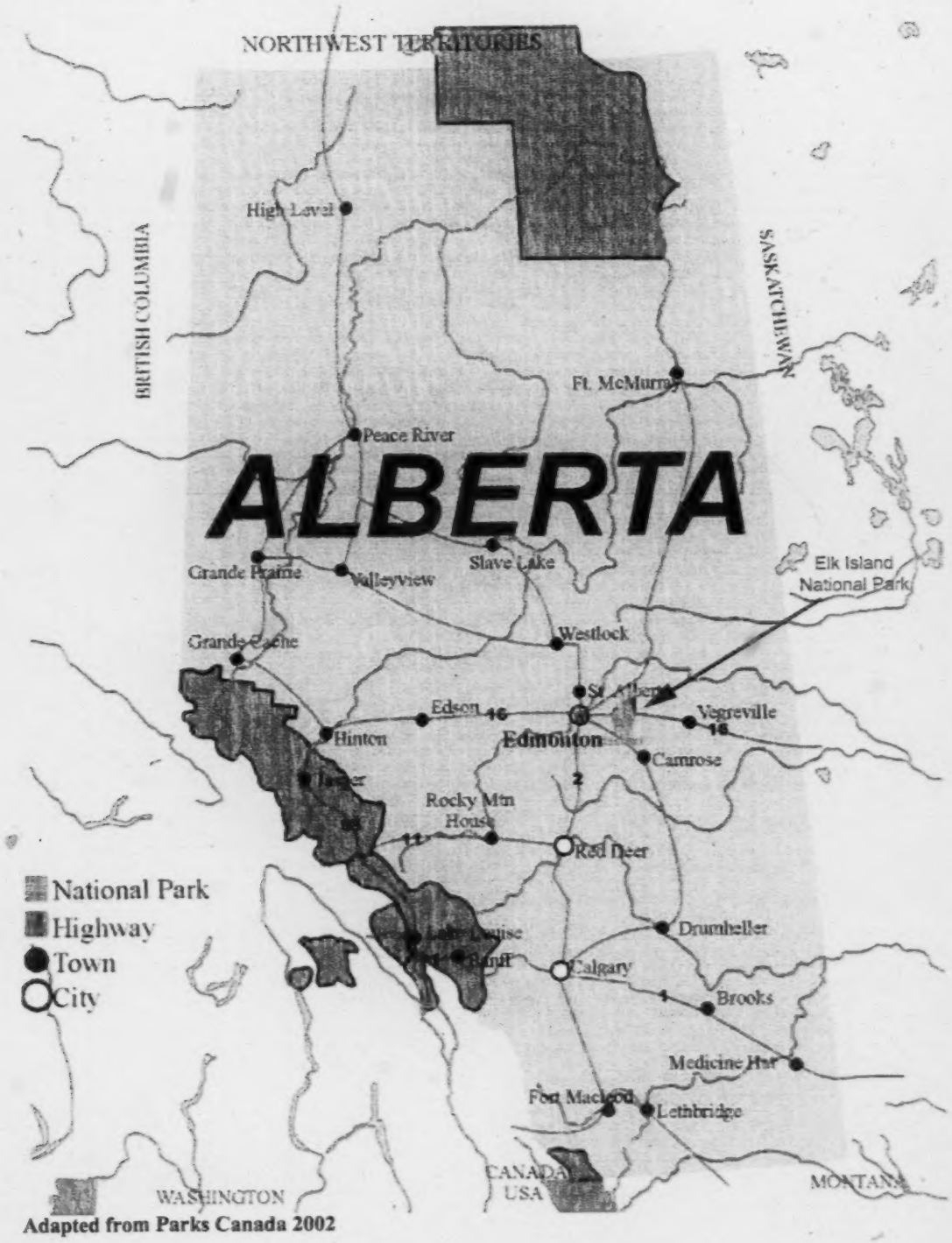
Dependent Variable	Treatment Interaction	Fencing Treatment	Seeding Treatment
Bare Ground	0.8223	0.4274	0.5518
Litter	0.3013	0.3709	0.1557
Live Vegetation	0.6612	0.2056	0.4909
Plant Density			
Seeded Species	0.5145	0.0121*	0.0729*
Non-Seeded Species	0.8079	0.2553	0.8421
Plant Biovolume	0.1710	0.1527	0.3966
Species Richness	0.1480	0.2777	0.1480

Where significance is set at p-values of 0.10

Table A16. Dependent variables and associated p-values for treatment interactions and main effects in 2000 at Elk Island National Park.

Dependent Variable	Treatment Interaction	Fencing Treatment	Seeding Treatment
Bare Ground	0.4673	0.0058*	0.4801
Litter	0.5047	0.0487*	0.6521
Live Vegetation	0.5251	0.1962	0.0829*
Plant Density			
Seeded Species	0.9065	0.2131	0.3423
Non-Seeded Species	0.3561	0.1896	0.4573
Plant Biovolume	0.2198	0.1484	0.4112
Biomass			
Fenced vs. Unfenced	0.6084	0.0107*	0.7927
Range Cage vs. Unfenced	0.9771	0.0044*	0.3614
Species Richness	0.3594	0.3760	0.4397

Where significance is set at p-values of 0.10



Adapted from Parks Canada 2002

Figure A1. Location of Elk Island National Park in Alberta.

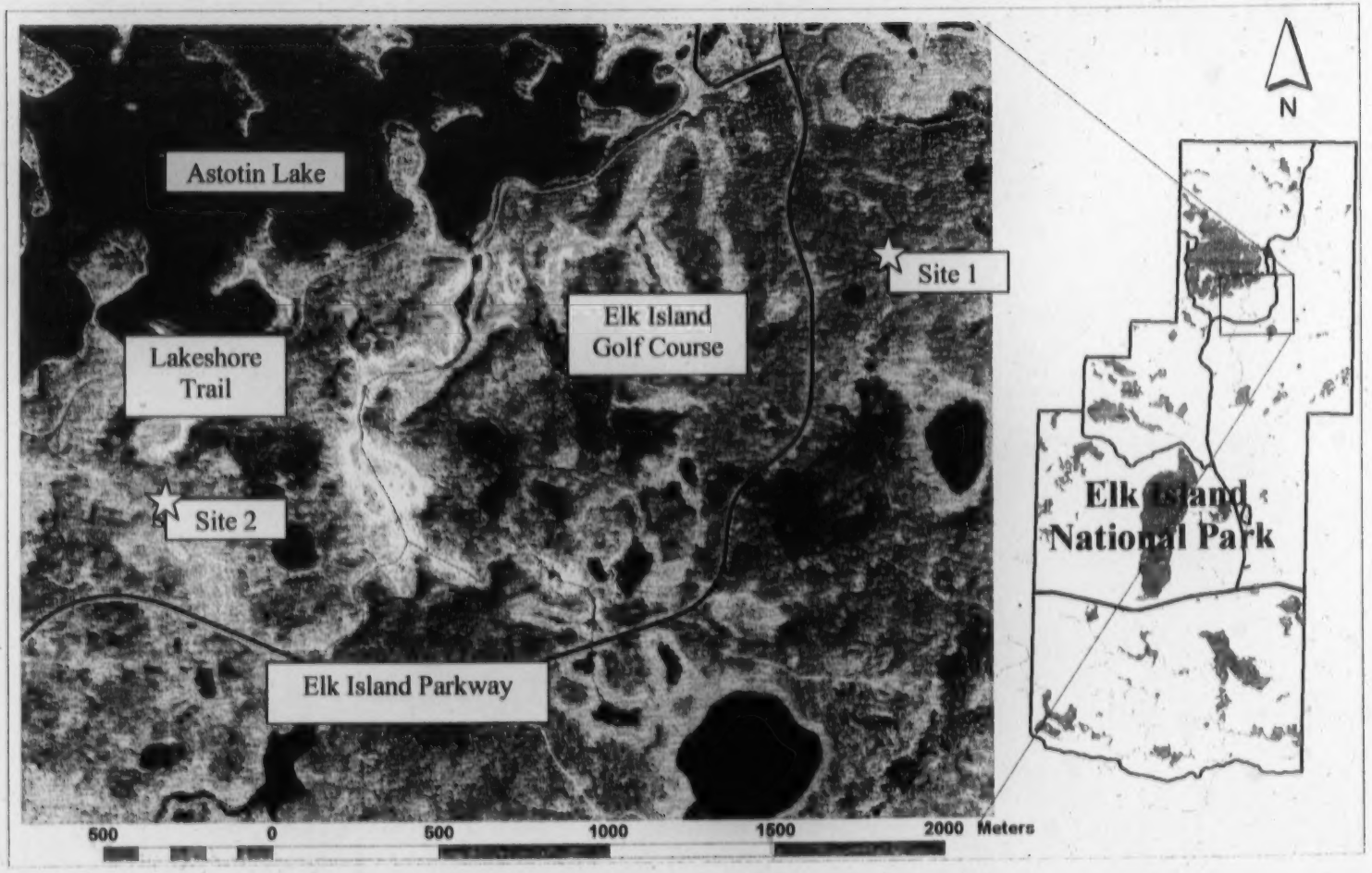


Figure A2. Location of Sites 1 and 2 in Elk Island National Park.

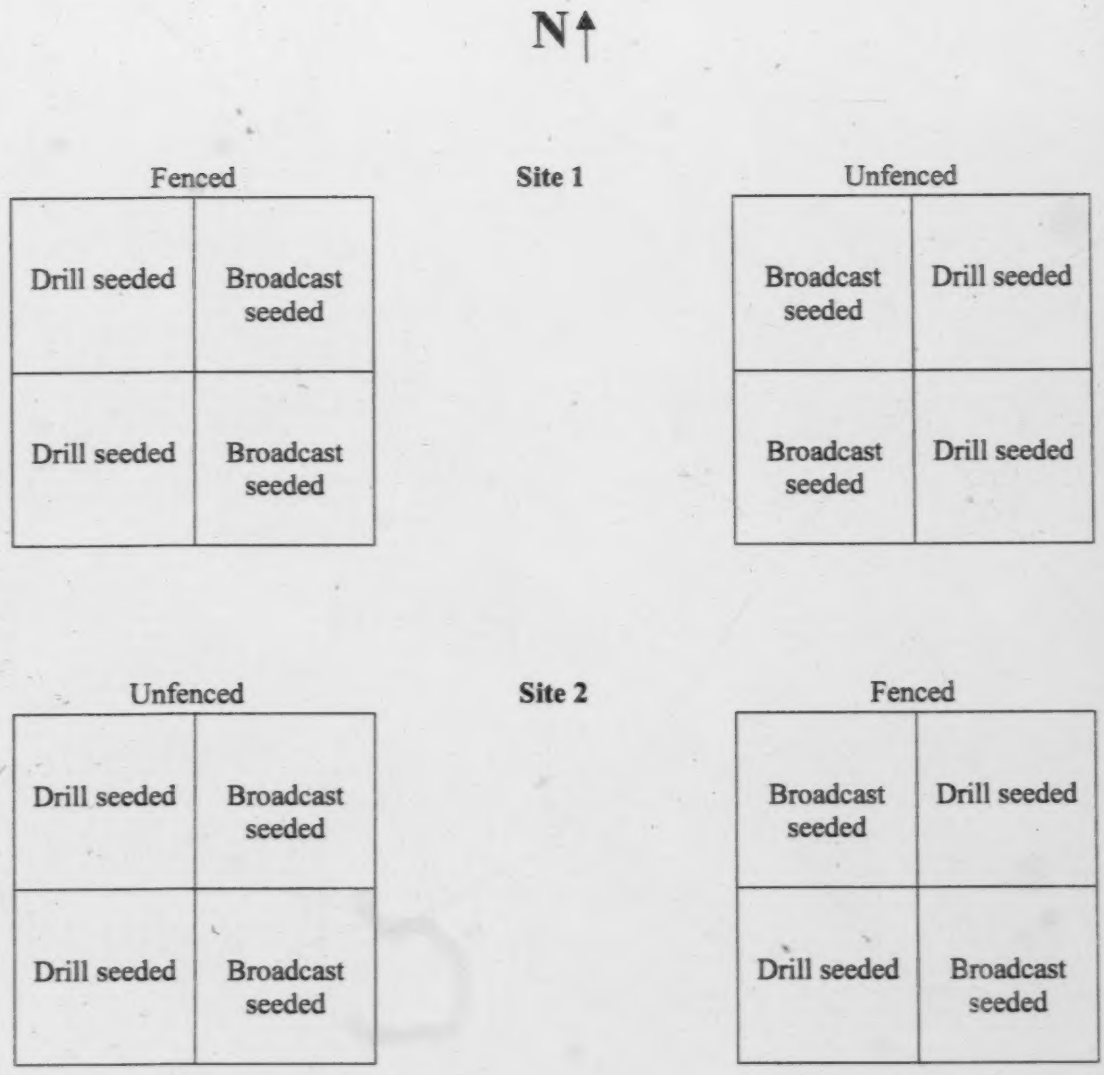


Figure A3. Site configuration of the split plot design incorporating randomized complete blocks.

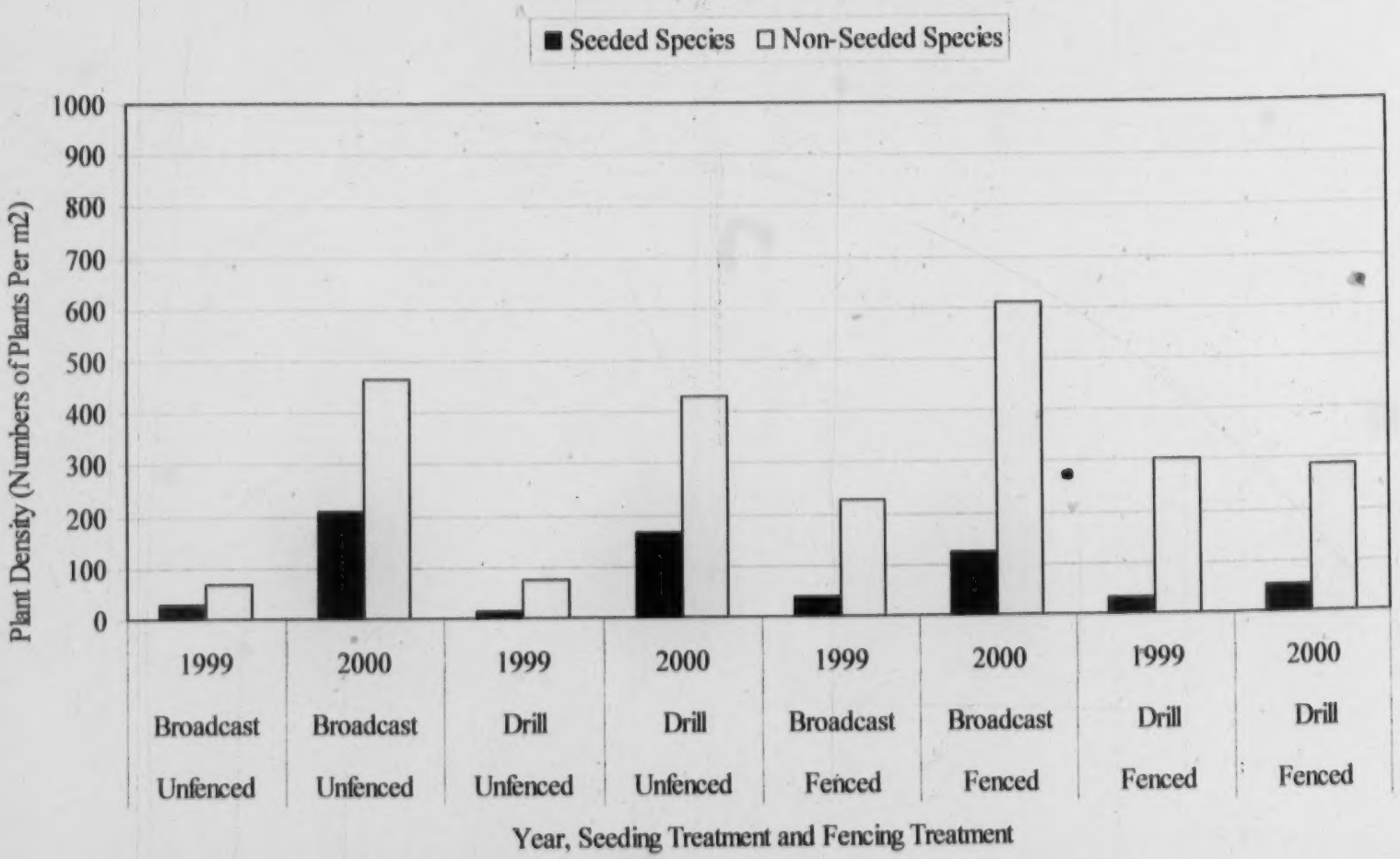


Figure A4. Mean plant density of seeded and non-seeded species for four treatment combinations in 1999 and 2000 at Site 1 in Elk Island National Park.

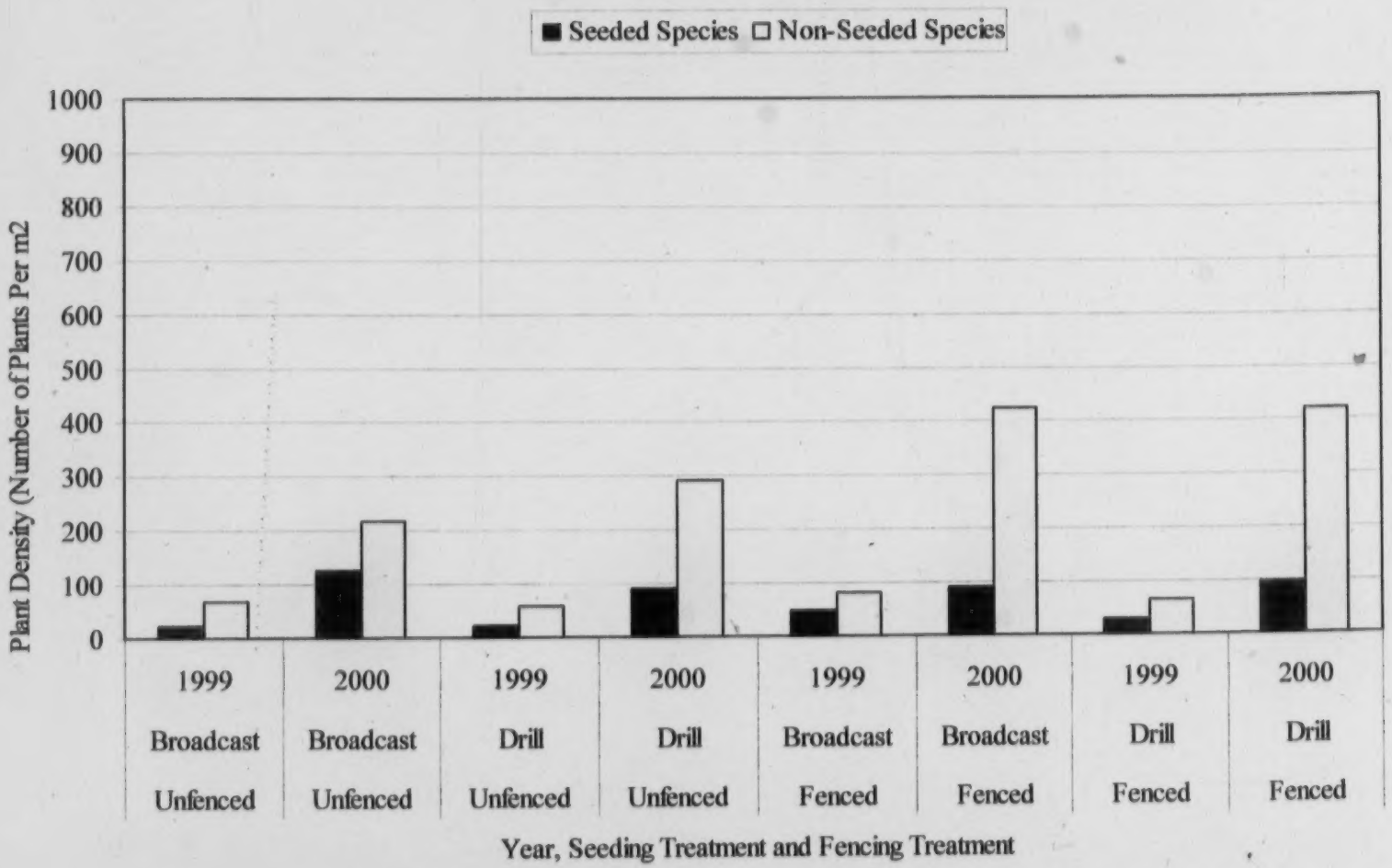


Figure A5. Mean plant density of seeded and non-seeded species for four treatment combinations in 1999 and 2000 at Site 2 in Elk Island National Park.

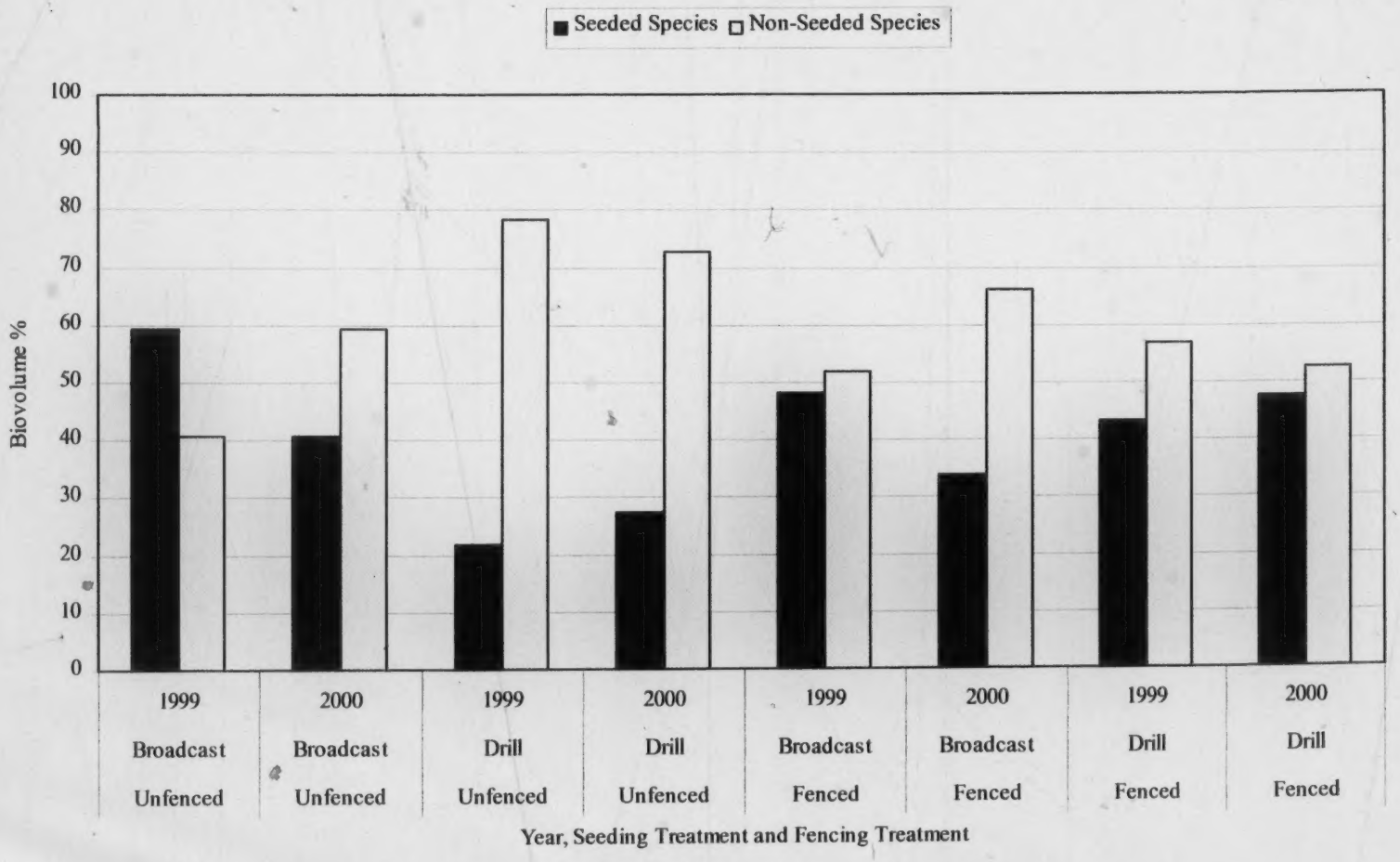


Figure A6. Mean plant biovolume of seeded and non-seeded species for four treatment combinations in 1999 and 2000 at Site 1 in Elk Island National Park.

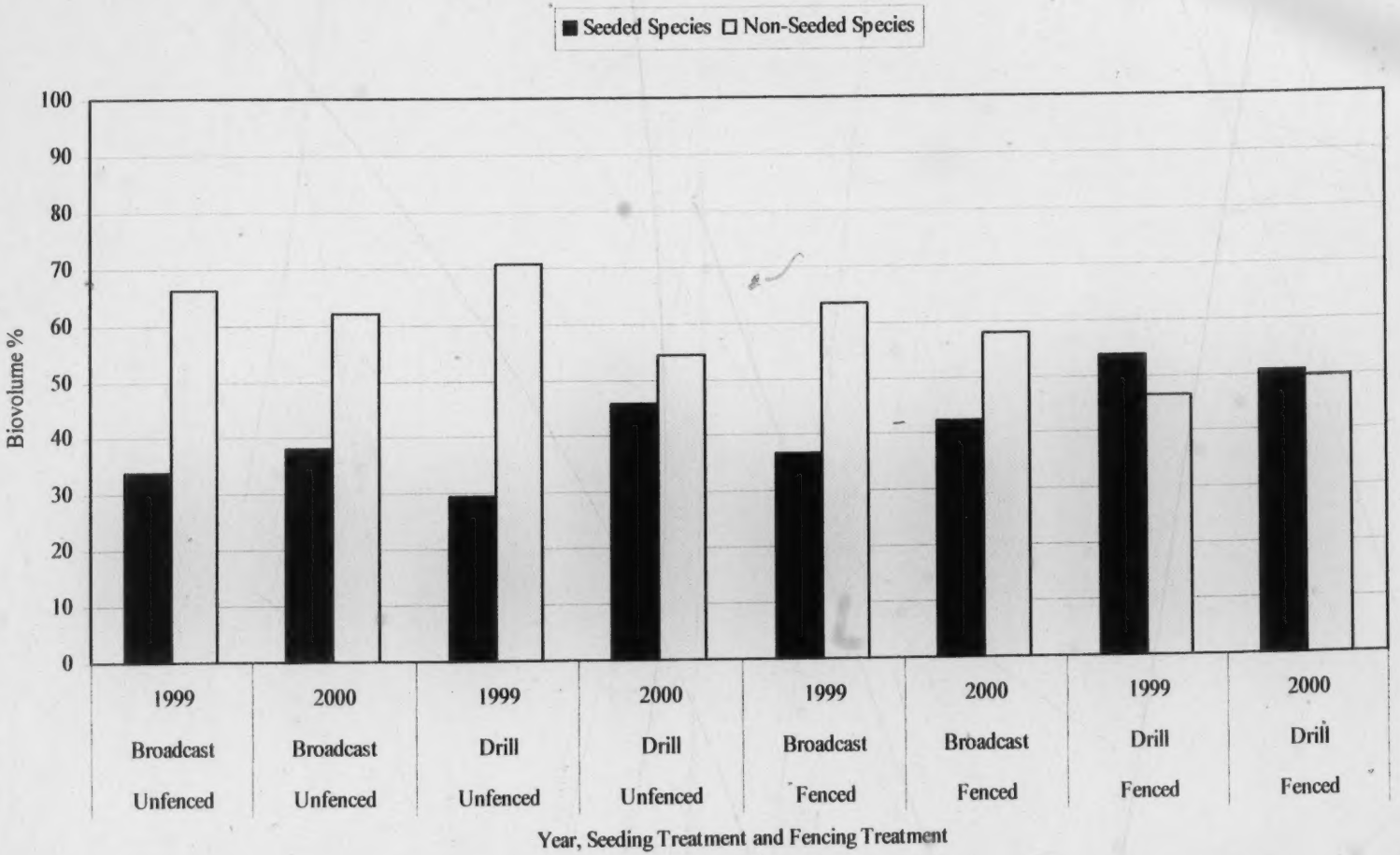


Figure A7. Mean plant biovolume of seeded and non-seeded species for four treatment combinations in 1999 and 2000 at Site 2 in Elk Island National Park.